UNIVERSITY OF CAPE TOWN

NITRIFICATION AND DENITRIFICATION KINETICS

in the

ACTIVATED SLUDGE PROCESS

by

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DECLARATION OF CANDIDATE

I, Adrianus Cornelius van Haandel, declare that this thesis is my own work and that it has not been submitted for a degree at another university.

Signed by candidate

May 1981

THESIS CONTRIBUTION TO KNOWLEDGE

The contribution of this investigation can be summarized as follows :

(1) The bi-substrate active-site death regeneration model, developed by Dold, Ekama and Marais (1980), to describe the aerobic activated sludge process,was extended to provide a reliable description of the behaviour of the nitrification-denitrification single sludge activated sludge process under constant, cyclic or unsteady conditions of flow and load. The extended model applies to series reactor systems containing aerated and non-aerated reactors and to single reactor systems having alternating aerated and non-aerated periods. The extension describing the behaviour in the anoxic state required no change in the basic equations describing the aerobic state. The numerical values of the kinetic constants in the anoxic state equations were found to be the same as those in the aerobic state equations, except for one: The value of the utilization rate constant for slowly biodegradable material in an anoxic environment is a fraction of approximately 0,38 of the value of this constant in an aerobic environment. Nitrification is affected in so far that growth of nitrifiers takes place in an aerobic environment, whereas death of nitrifiers takes place in both aerobic and anoxic environments.

(2) The existing empirical denitrification models (valid for constant flow and load conditions) could be deduced from the solutions of the basic equations. The empirical denitrification rate constants could be interpreted in terms of the expressions for utilization of easily and slowly biodegradable organic substrates in an anoxic environment. These constants were shown to have no fundamental kinetic significance; they are only apparent constants, the result of kinetic reactions which, fortuitously, show little variation under normal operational conditions.

(3) It was established that some of the basic kinetic constants that determine the rates of nitrification and denitrification in a single sludge activated sludge process depend upon the influent sewage source and should be determined experimentally for each waste flow. The constant particularly dependent on the waste flow source is the maximum specific growth rate of nitrifiers; the utilization rate constant for slowly biodegradable material is also affected but to a lesser degree. To establish the value of these two constants experimentally, a convenient set up is the single reactor completely mixed activated sludge process in which the mixed liquor is submitted to alternating anoxic and aerobic periods.

(4) The alkalinity changes the influent undergoes as it passes through the activated sludge process, due to nitrification, denitrification, ammonification and de-ammonification, can readily be incorporated in the general model. These alkalinity changes can be predicted so accurately that changes in alkalinity of the mixed liquor in a single reactor alternating anoxic and aerobic system can be used as sensitive parameters to determine the rates of nitrification and denitrification. Also it was shown theoretically that, when the mixed liquor alkalinity drops below 35 $mgCaCO_3.\ell^{-1}$, the pH will become unstable. This is of particular importance in the design of processes for low alkalinity waste streams.

(5) A design procedure was developed to optimize single sludge nitrification - denitrification systems, i.e. to maximize nitrogen removal in these systems. It was shown that for a particular sludge age and temperature, the extent of nitrification or nitrification capacity is linearly related to the influent TKN concentration, N_{ti}, and the extent of denitrification achievable or denitrification potential is proportional to the influent COD concentration, S_{ti}. The effluent nitrate concentration depends upon the relative magnitudes of the

nitrification capacity and the denitrification potential and hence on the N_{ti}/S_{ti} ratio. Depending on this ratio three different situations may arise :

(1) Low N_{ti}/S_{ti} ratio : in this case complete nitrate removal is possible by having a series reactor system with anoxic reactors before and after the main aerated reactor (i.e. by having a pre- and a post-denitrification reactor).

(2) Medium N_{ti}/S_{ti} ratio : in this case complete nitrate removal is not possible; maximum nitrate removal is obtained in a system with both pre- and post-denitrification reactors.

(3) High N_{ti}/S_{ti} ratio : in this case complete nitrate removal is not possible; maximum nitrate removal is obtained in a pre-denitrification system.

Expressions were developed to optimize the operational parameters (reactor configuration, sludge age, recycle ratio) for maximum nitrogen removal as a function of the sewage characteristics and kinetic constants. Sewage characteristics (besides the N_{ti}/S_{ti} ratio) that influence markedly the optimal values of the operational parameters are the composition of the influent COD and the temperature. Kinetic constants that have a marked influence are the maximum specific growth rate of nitrifiers and the utilization rate constant of slowly biodegradable material in an anoxic environment.

From the work presented in this thesis it would appear that a design method is now available for single sludge nitrification-denitrification systems that allows a rational selection of the operational parameters to affect maximum nitrogen removal for any waste flow with specified values for sewage characteristics and kinetic constants.

SYNOPSIS

The objectives of this investigation were (1) to develop a general model for nitrification and denitrification kinetics in single sludge activated sludge systems and (2) to develop a design procedure for maximum nitrogen removal from waste flows treated in these systems.

Empirical models for denitrification in the single sludge activated sludge process have all been based on observed behaviour in batch or continuous flow reactors. Marais and several co-workers conducted a particularly intensive investigation into the denitrification behaviour in single sludge systems by using anoxic plug flow reactors under constant flow and load conditions. From their observations they concluded that the denitrification rate is (1) constant with time (2) independent of the nitrate concentration (3) apparently proportional to the active sludge concentration and (4) independent of sludge age for sludge ages between 10 and 20 days. However, the observed denitrification rate constants depended upon the position of the anoxic reactor. In a predenitrification reactor (an anoxic reactor receiving the influent flow) two rates were identified: A short primary phase with a high rate of denitrification and a secondary phase with a lower rate of denitrification, which continued for the balance of the time in the reactor. The extent of denitrification due to the high rate of denitrification was found to be proportional to the influent COD concentration. In a post-denitrification reactor (an anoxic reactor receiving nitrified mixed liquor from a preceding aerobic reactor) a single phase was observed, with a denitrification rate lower than in the secondary phase of the predenitrification reactor. Marais and his co-workers proposed the following empirical equation for the system nitrate removal in a single sludge system with a pre- and a post-denitrification reactor under constant flow and load conditions:

$$\Delta N_s = \alpha.COD_{inf} + K_2 X_a (R_1 - R') + K_3 X_a R_3 \qquad (i)$$

where

ΔN_s = system nitrate removal (mg NO_3-N.ℓ^{-1})

COD_{inf} = influent COD concentration (mg COD.ℓ^{-1})

K_2, K_3 = denitrification rate constants in a pre- and post-denitrification reactor respectively (mg NO_3-N.mg $X_a^{-1}h^{-1}$)

R_1, R_3 = nominal retention times in a pre- and post-denitrification reactor respectively (h)

R' = duration of the primary phase (h)

X_a = active sludge concentration (mg VSS.ℓ^{-1})

α = proportionality constant (mg NO_3.mg COD^{-1})

This equation was found to model nitrate removal very well but no clear link with the basic kinetics of the activated sludge process was apparent. In order to enquire if such a link could be established, the hypothesis was made that the behaviour of heterotrophic organisms in an anoxic environment is not qualitatively different from that in an aerobic environment. This hypothesis did not seem to be unreasonable because the metabolic pathway of degradation of organic material in an anoxic environment is not fundamentally different from that in an anoxic environment and is mediated by the same micro-organisms. A first step to verify the validity of this hypothesis was to simulate the deoxygenation behaviour in aerobic plug flow reactors using the general model for the aerobic activated sludge process presented by Dold, Ekama and Marais (1980). The simulated deoxygenation behaviour in aerobic plug flow reactors was remarkably similar to the observed denitrification behaviour in anoxic plug flow reactors. It was accepted therefore, that the aerobic model of Dold *et al* formed a basis for describing denitrification behaviour.

A quantitative correspondence between simulated and experimentally observed denitrification behaviour in anoxic plug flow reactors under constant flow and load conditions was obtained by changing the numerical value of one of the kinetic constants proposed by Dold *et al* for the aerobic activated sludge process and making some secondary changes required to incorporate other characteristics of anoxic reactors, namely, that nitrate instead of oxygen acts as an electron-acceptor and that in these reactors no nitrification takes place.

By this approach it was possible to interpret the empirical constants α, K_2 and K_3, in terms of a general kinetic theory for single sludge nitrification-denitrification activated sludge processes. The empirical constant α is proportional to the concentration of easily biodegradable influent COD; the constants K_2 and K_3 are associated with the constant of substrate utilization of slowly biodegradable material. The model thus constructed and calibrated was successfully applied to simulate the behaviour of nitrification and denitrification in all the processes investigated. These included series reactor systems, both under constant and under cyclic flow and load conditions, and single reactor systems in which the mixed liquor was placed alternatively in an anoxic and in an aerobic environment.

In the course of the investigation it became apparent that the influent sewage characteristics had a dual influence on the nitrification and denitrification behaviour:

(i) the concentration and composition of the influent TKN and COD influenced the extent of nitrification and denitrification respectively.

(ii) some kinetic constants assumed different values for different sewage sources.

The kinetic constants that seemed to be affected most severely by the characteristics of the influent were the maximum specific growth rate of nitrifiers and the utilization rate constant for slowly biodegradable material. Because the exact nature of the sewage characteristics that caused the variability of the kinetic constants could not be established, the need arose to develop a system that would allow accurate, reliable and rapid experimental assessment of these constants and hence, of the nitrification and denitrification behaviour in anoxic/aerobic activated sludge systems. The system that proved to comply to a maximum degree with these requirements was the single reactor completely mixed activated sludge process, operated under constant flow and load conditions, in which the mixed liquor was submitted to alternating periods of anoxic and aerobic environments. An additional advantage of this system was that it was found to be possible to calculate approximated values of the kinetic constants for nitrification and denitrification from observed nitrate concentration-retention time profiles

manually,without the aid of computer simulations.

Using the single reactor system with alternating anoxic and aerobic periods,it was possible to show that changes in the mixed liquor alkalinity also could be interpreted in terms of the general model. Excellent correlation between predicted and observed alkalinities were obtained by assuming that the alkalinity of the mixed liquor was affected by nitrification, denitrification, ammonification and de-ammonification and that the effects of these reactions on alkalinity were in conformity with the theoretical stoichiometry. Hence alkalinity-time profiles in the single reactor system could also be used for calculations of the kinetic constants of nitrification and denitrification.

Whereas the general model describes the kinetic behaviour of any single sludge nitrification denitrification process, it does not provide information concerning the optimal values of operational parameters to affect maximum nitrogen removal in a particular design situation. For the purpose of maximization of nitrogen removal an optimization procedure was developed that allows a rational selection of all the operational parameters (recycle ratios, sludge age, size and division of the anoxic sludge mass fraction) for specified values of relevant sewage characteristics and kinetic constants . The optimal design procedure was developed for constant flow and load conditions. However, from simulations using the general model,it would appear that the optimal solution for constant flow and load conditions is also near optimal for cyclic flow and load conditions.

The basic considerations on which the optimization procedure is based are:

(i) Nitrification must be efficient (>95 per cent)

(ii) Denitrification must be complete or maximum.

(i) Nitrification.

Nitrifiers are obligate aerobes, i.e. growth of nitrifiers and nitrification takes place in an aerobic environment only. Death of nitrifiers, however, takes place in both aerobic and anoxic environments. Hence, in order to maintain a concentration of nitrifiers sufficiently high for efficient nitrification, there is a minimum aerobic sludge mass

fraction. From a theoretical analysis this minimum aerobic sludge mass fraction was found to depend on several factors: (1) the required nitrification efficiency (i.e. the maximum allowable effluent ammonia concentration), (2) the maximum specific growth rate of nitrifiers, (3) the mixed liquor temperature, and (4) the operational sludge age.

Also, from experimental observations (Arkley and Marais, 1981), if the aerobic sludge mass fraction decreases below 50 per cent of the total sludge mass there is a rapid deterioration of sludge settling characteristics. Hence, in practical nitrification-denitrification systems the minimum aerobic sludge mass fraction is constrained by the requirements for efficient nitrification and sludge settleability.

(ii) Denitrification.

The minimum aerobic sludge mass fraction required for efficient nitrification and sludge settleability sets a maximum to the anoxic sludge mass fraction that can be in an anoxic environment. The extent of denitrification was formulated in terms of Eq (i) and depends upon the following factors: (1) the influent biodegradable COD concentration and the magnitude of the easily biodegradable fraction, (2) the mixed liquor temperature, (3) the operational sludge age, (4) the utilization rate constant for slowly biodegradable material, and (5) the magnitude of the anoxic sludge mass fraction and the division of this fraction over the pre- and post-denitrification reactor.

In the development of the optimization procedure it became apparent that there are two factors that most strongly influence the optimal values of the operational parameters: the maximum specific growth rate of nitrifiers and the influent TKN/COD ratio. In broad outline the maximum specific growth rate of nitrifiers determines the anoxic sludge mass fraction and the sludge age. The influent TKN/COD ratio governs the process configuration: at low influent TKN/COD ratios complete nitrate removal is possible, using an appropriate system with pre- and post-denitrification reactors. At high influent TKN/COD ratios nitrate removal is incomplete and the optimal configuration is a pre-denitrification system.

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LIST OF SYMBOLS

Symbol	Description
a	recycle ratio from an aerobic to a predenitrification reactor
b_h	endogenous respiration rate constant for active sludge (d^{-1})
b'_h	death rate constant for active sludge (d^{-1})
b_n	death rate constant for nitrifiers (d^{-1})
f	inert residue in the endogenous respiration approach
f'	inert residue in the death regeneration approach
f_{ca}	ratio easily : total biodegradable material in the influent
f_M	maximum anoxic sludge fraction allowing efficient nitrification
f_m	anoxic sludge mass fraction causing cessation of nitrification
f_{max}	maximum anoxic sludge mass fraction set by requirements for settleability
f_{min}	minimum anoxic sludge mass fraction for complete utilization of easily biodegradable material
f_n	mass fraction of nitrogen in organic sludge (mg N.mg VSS^{-1})
f_{na}	ratio ammonia : total TKN in the influent
f_{ns}	ratio TKN : organic material in stored material (mg N.mg COD^{-1})
f_{nu}	unbiodegradable soluble TKN fraction in the influent
f_p	mass fraction of phosphorus in organic sludge (mg P.mg VSS^{-1})
f_{up}	unbiodegradable particulate influent COD fraction (mg VSS.mg COD^{-1})
f_{us}	unbiodegradable soluble influent COD fraction
f_x	anoxic sludge mass fraction
k_1	first dissociation constant of CO_2 in water
k_2	second dissociation constant of CO_2 in water

m_{oc} mass of oxygen consumed for carbonaceous material degradation per unit daily applied COD

m_x mass of sludge present in an activated sludge system per unit of daily applied COD (additional indeces a, e,i and v refer to active sludge, endogenous residue, inert material and volatile solids respectively).

q sludge wastage flow ($\ell.d^{-1}$)

r general symbol indicating the rate of a particular mechanism or the reaction rate of one of the variables in the activated sludge process (all in $mg.\ell^{-1}.d^{-1}$)

(1) mechanisms.

index a refers to adsorbtion of slowly biodegradable material

" ala " " alkalinity due to ammonification

" aln " " alkalinity due to nitrification

" am " " ammonification

" co " " conversion of organic nitrogen to ammonia

" d " " death of active sludge

" den " " denitrification

" dn " " death of nitrifiers

" dr " " net effect of death and regeneration

" e " " formation of endogenous residue in the endogenous respiration approach

" er " " endogenous respiration

" fe " " formation of endogenous residue in the death regeneration approach

" g " " growth of active sludge

" gn " " growth of nitrifiers

" hy " " hydrolysis of stored material

" n " " nitrification

" nd " " change of organic nitrogen concentration due to desorbtion

" nh " " synthesis of ammonia in active sludge

index nl refers to production of organic nitrogen due to lysis

" ns " " change of organic nitrogen concentration due to adsorbtion

" o " " oxygen uptake for carbonaceous material oxidation

" on " " oxygen uptake for nitrification

" p " " internal generation of slowly biodegradable material due to lysis of active sludge

" r " " regeneration of active sludge

" syn " " synthesis of active sludge

" us " " utilization of easily biodegradable material

" up " " utilization of slowly biodegradable material

(2) variables.

index alk refers to alkalinity

" na " " ammonia concentration

" nn " " nitrate concentration

" no " " organic nitrogen concentration

" sbp " " particulate biodegradable material concentration

" sbs " " easily biodegradable material concentration

" su " " soluble unbiodegradable material concentration

" xa " " active sludge concentration

" xe " " endogenous residue concentration

" xi " " inert material concentration

" xn " " nitrifier concentration

" xs " " stored material

" xv " " volatile solids

r_{Cj}	rate of change of variable C in reactor j due to both reaction and hydraulic effects
r_{Cjh}	rate of change of variable C in reactor j due to hydraulic effects
r_{Cjr}	rate of change of variable C in reactor j due to reaction effects
s	underflow recycle ratio
t_p	duration of the primary phase in a predenitrification reactor (h)
w	general symbol indicating the wastage of sludge or nutrient in wasted sludge per unit applied COD
w_n	nitrogen wasted with sludge per unit influent COD (mg N.mg COD^{-1})
w_p	phosphorus wasted with sludge per unit influent COD (mg P.mg COD^{-1})
w_{xa}	active sludge wastage per unit influent COD (mg Xa.mg COD^{-1})
w_{xe}	endogenous residue wastage per unit influent COD (mg VSS.mg COD^{-1})
w_{xi}	inert sludge wastage per unit influent COD (mg VSS.mg COD^{-1})
w_{xv}	volatile sludge wastage per unit influent COD (mg VSS.mg COD^{-1})
A	fractional amplitude of a sinoidal wave (subscripts f, COD and TKN refer to influent flow, COD concentration and TKN concentration respectively)
alk	mixed liquor alkalinity (ppm $CaCO_3$)
C_e	concentration of a variable C (mg.ℓ^{-1})
	subscript e refers to effluent subscript i refers to influent subscript j refers to reactor j subscript n refers to reactor n
C_r	sludge mass factor = active sludge mass in the system per unit daily utilized COD (mg X_a.mg COD^{-1} d^{-1})
D_c	denitrification capacity (mg NO_3-N.ℓ^{-1})
DO	dissolved oxygen concentration (mg O.ℓ^{-1})
D_o	system oxygen removal in an anoxic reactor (expressed as mg equivalent nitrate per liter influent)

D_p denitrification potential (mg NO_3-N.ℓ^{-1})

F_d activity coefficient for a divalent ion

F_m activity coefficient for a monovalent ion

F_{ma} maximum ratio stored material/active sludge

K empirical denitrification rate constant (mg N.mg VSS.$^{-1}h^{-1}$)

K' empirical denitrification rate constant (mg N.mg $X_a^{-1}.h^{-1}$)

K_1 high rate (primary phase) denitrification rate constant (mg N.mg X_a^{-1} h^{-1})

K_2 low rate (secondary phase) denitrification rate constant (mg N.mg X_a^{-1} h^{-1})

K_3 denitrification rate constant in a postdenitrification reactor (mg N.mg X_a^{-1} h^{-1})

K_{mp} substrate utilization rate constant for slowly biodegradable material in an aerobic environment (mg COD.mg $X_a^{-1}.d^{-1}$)

K_{mp}^1 substrate utilization rate constant for slowly biodegradable material in an anoxic environment (mg COD.mg $X_a^{-1}.d^{-1}$)

K_{ms} substrate utilization rate constant for easily biodegradable material (mg COD.mg X_a^{-1} d^{-1})

K_n half saturation value for nitrification (mg NH_3-N.ℓ^{-1})

K_{ns} substrate utilization rate constant for nitrification (mg NH_3-N.mg $X_n^{-1}.d^{-1}$)

K_o half saturation value for limiting oxygen in nitrification (mg O.ℓ^{-1})

K_r conversion rate constant for organic nitrogen (ℓ.mg X_a^{-1} d^{-1})

K_{sp} half saturation value for slowly biodegradable material (active site approach) (mg COD.mg X_a^{-1})

K_{ss} half saturation value for easily biodegradable material (mg COD.ℓ^{-1})

K_v adsorption rate constant for adsorption of particulate biodegradable material (ℓ.mg $X_a^{-1}.d^{-1}$)

L_{cj} load of variable C on reactor j (mg.d^{-1})

M_{Cjh} mass loading rate of reactor j due to net input and output flows (mg.d^{-1})

MO mass of oxygen required daily in an activated sludge system (mg O.d^{-1})

index c refers to carbonaceous material oxidation

" n " " nitrification

MS daily mass of COD to or from an activated sludge process (mg COD.d^{-1})

index sb refers to biodegradable material

" t " " total COD

MX mass of sludge present in an activated sludge system (mg VSS)

index a refers to active sludge

" e " " endogenous residue

" i " " inert material

" n " " nitrifiers

" v " " volatile solids

N general parameter indicating concentration of nitrogenous material (mg N.ℓ^{-1})

index a refers to ammonia concentration

" a,co " " effluent ammonia (constant flow and load)

" a,cy " " effluent ammonia concentration (cyclic flow and load)

" ad " " desired effluent ammonia concentration

" ae " " nitrate concentration in the aerobic reactor

" av " " nitrate concentration available for denitrification

" c " " nitrification capacity

" in " " influent nitrate concentration

" n " " nitrate concentration

" ne " " effluent nitrate concentration

" o " " organic nitrogen concentration

" p " " nitrification potential

" pi " " influent TKN concentration associated with particulate unbiodegradable material

" s " " influent TKN concentration required for sludge production

index sm refers to minimum R_s for nitrification

" sm' " " minimum R_s for efficient nitrification

S general symbol indicating COD concentration (mg COD.ℓ^{-1})

index b refers to biodegradable material

" bp " " slowly biodegradable material

" bs " " easily biodegradable material

" t " " total COD

" u " " unbiodegradable soluble material

S_f safety factor for efficient nitrification

T temperature (°C)

V total reactor volume (ℓ)

V_{an} anoxic reactor volume (ℓ)

X general symbol indicating sludge concentration (mg VSS.ℓ^{-1})

index a refers to active sludge

" ao " " active sludge (initial)

" at " " active sludge (after time t)

" e " " endogenous residue

" i " " inert material

" ii " " inert influent material

" n " " nitrifiers

" s " " stored material

" v " " volatile solids

Y general symbol indicating yield coefficient

index h refers to yield of active sludge (mg VSS.mg 100^{-1})

" n " " yield of nitrosomonas (mg VSS.mg N^{-1})

" ni " " yield of nitrobacter (mg VSS.mg N^{-1})

α proportionality constant between extent of high rate nitrate removal and influent biodegradable COD concentration (mg NO_3-N.mg COD^{-1})

$= f_{ca}(1-PY_h)/2,86$

ΔN_a	reactor nitrate removal in an anoxic reactor (mg $N.\ell^{-1}$)
ΔN_s	system nitrate removal in an anoxic reactor (mg $N.\ell^{-1}$)
η	ratio of substrate utilization constants for slowly biodegradable material in an anoxic and in an aerobic environment, K^1_{mp}/K_{mp}
θ	temperature dependency coefficient (simplified Arrhenius equation)
μ_n	specific growth rate of nitrifiers (d^{-1})
μ_{nm}	maximum specific growth rate of nitrifiers (d^{-1})
μ'_{nm}	apparent maximum specific growth rate of nitrifiers in anoxic/aerobic systems (d^{-1})

additional subscript e refers to effluent concentration

" " i " " influent concentration

" " T " " temperature (°C)

" " 1 " " pre-denitrification

" " 3 " " post-denitrification

CHAPTER ONE

INTRODUCTION

Single sludge nitrification-denitrification processes were introduced in South Africa through the work of Barnard (1973). He combined the Ludzack-Ettinger process (Ludzack and Ettinger, 1962) with that of Wuhrmann (Wuhrmann, 1956) and introduced a system of controlled recycles to give what is now known as the Bardenpho process. The Bardenpho configuration has found wide application in South Africa particularly in its extension to remove phosphorus by a biological excess mechanism, called the Phoredox system (Barnard,1975).

With regard to design of these systems a major difficulty in application has been the lack of a consistent kinetic model to describe the complex systems of reactions that take place in the process. Difficulties in developing such a model are indeed formidable: the system consists of a series of reactors, some of which are aerated and some unaerated; energy (COD, BOD) is removed in both aerated and unaerated reactors; efficient nitrification by obligate autotrophic organisms must be ensured in an environment that is alternately aerobic and anoxic; nitrate removal can take place only in unaerated reactors and the rate of nitrate removal differs,depending on the position of the reactor in the series.

Design information for these systems was obtained from laboratory and pilot scale studies, either on the specific waste flow or by empirical extension of the available data to the waste flow under consideration.

The first attempt at developing a theory to describe denitrification kinetics was limited by formulating denitrification as a zero order reaction with respect to nitrate, as follows:

$$\Delta N/\Delta t = KX_v \qquad (1.1)$$

where

$\Delta N/\Delta t$ = denitrification rate (mg $N.\ell^{-1}h^{-1}$)

X_v = volatile suspended solids concentration (mg $VSS.\ell^{-1}$)

K = denitrification rate constant (mg $N.mg\ VSS^{-1}h^{-1}$)

Values of K were obtained from batch tests and from completely mixed anoxic reactors in series reactor systems. One difficulty observed was that widely different values for K were obtained for different operational conditions. Hence this approach could only be used reliably if the operational conditions of the system to be designed conformed closely to those of the system on which the K value was determined. There was no apparent link between the empirical constant K and the activated sludge kinetic theory then extant.

In 1971 Marais and his group initiated a wide ranging enquiry into activated sludge kinetics. By combining and extending the models of McKinney (1962) and of Lawrence and McCarty (1970) they developed a consistent kinetic theory which described very well the behaviour of the aerobic activated sludge process under constant flow and load conditions. They also investigated the Bardenpho system, replacing completely mixed reactors by plug flow reactors and found that Eq (1.1) should be reformulated in terms of concentration of active sludge, X_a, instead of volatile sludge, X_v, i.e. :

$$\Delta N/\Delta t = K'X_a. \qquad (1.2)$$

The concentration of active sludge was estimated by means of the formulation developed for aerobic systems. In the reformulated expression (Eq (1.2)) the observed K' values depended upon the position of the anoxic reactor. In a predenitrification reactor (i.e. an anoxic reactor receiving the influent flow) there were two denitrification phases: a primary phase, lasting only for a short period during which a high denitrification rate was observed, followed by a secondary phase with a lower rate. In a post-denitrification reactor (i.e. a reactor receiving mixed liquor from an aerobic reactor) a single denitrification phase with constant rate was observed. All the phases, when formulated in terms of Eq (1.2) gave respective denitrification rate constants, K', that

remain constant over the sludge age range 10 to 20 days and that are dependent only on temperature. These experimental observations allowed the following formulation for the nitrate removal in a process with both a pre- and a post-denitrification reactor:

$$\Delta N_s = K'_1 X_a R' + K_2 X_a (R_1 - R') + K_3 X_a R_3 \qquad (1.3)$$

where

ΔN_s = system nitrate removal (mg N.ℓ^{-1})

R' = duration of the primary phase (nominal) (h)

R_1 = nominal retention time in the predenitrification reactor (h)

R_3 = nominal retention time in the post-denitrification reactor (h)

K'_1 = denitrification rate constant during the primary phase in a predenitrification reactor (mg N.mg X_a^{-1}.h^{-1})

K_2 = denitrification rate constant during the secondary phase in a predenitrification reactor (mg N.mg $X_a^{-1}h^{-1}$)

K_3 = denitrification rate constant in a post-denitrification reactor (mg N.mg $X_a^{-1}h^{-1}$)

The difficulty with this formulation was that it was necessary to evaluate R'. However, Marais and his co-workers established that the mass concentration of nitrate removed in the primary phase was proportional to the influent COD concentration. Consequently R' could be estimated and Eq (1.3) could be rewritten as:

$$\Delta N_s = \alpha.COD_{inf} + K_2 X_a (R_1 - R') + K_3 X_a R_3 \qquad (1.4)$$

where

α = proportionality constant (mg N.mg COD^{-1})

COD_{inf} = influent COD concentration (mg COD.ℓ^{-1})

This formulation was successfully applied to describe nitrate removal in completely mixed reactors. However Marais and his co-workers could not establish a link between their empirical expression (Eq (1.4)) and activated sludge theory.

While pursuing the experimental investigation into anoxic/aerobic systems work was undertaken to further develop the aerobic model so as to describe the behaviour under cyclically varying flow and load conditions. In order to achieve this objective it was found necessary to reconsider the basis for the existing theory of the aerobic activated sludge process. The final conceptional basis on which the general model was developed can be summarized as follows (Dold, Ekama and Marais, 1980):

1. The influent biodegradable material in municipal waste flows consists of two fractions: (a) an easily biodegradable, soluble fraction that is directly and rapidly metabolized, and (b) a slowly biodegradable particulate fraction that requires adsorption onto the sludge and extracellular hydrolysis prior to absorption and metabolism. The rate of utilization of easily biodegradable material is described by the Monod Equation (Monod, 1950). The rate of utilization of slowly biodegradable material is determined by the rate of hydrolysis. As hydrolysis occurs at the surface of the active organisms, the rate of hydrolysis and hence of utilization of slowly biodegradable material is formulated in terms of the active site theory of Levenspiel (1972).

2. The classical synthesis-endogenous respiration approach is inadequate to describe the behaviour of the organisms and needs to be replaced by a death-regeneration approach.

3. With regard to nitrification, ammonification and deammonification need to be added to the classical theory of Downing (Downing, Painter and Knowles, 1964).

Using the general model, it was found possible to describe the behaviour of the aerobic activated sludge process over a wide range of process configurations, sludge ages and temperatures.

In anoxic activated sludge reactors nitrate (or nitrite) replaces oxygen as electron acceptor, for the oxidation of biodegradable material. Both reactions are mediated by the same heterotrophic

organisms and the degradation reaction of biodegradable material, using either nitrate or oxygen as an electron acceptor, follows very similar pathways. This raised the possibility that perhaps substrate oxidation in an anoxic environment and the associated denitrification reaction in single sludge processes could be integrated into the aerobic theory; this investigation describes endeavours towards attaining this objective.

In Chapter 3 the aerobic theory is thoroughly reviewed as this theory forms the basis for the extension to denitrification. In Chapter 4 it will be shown that denitrification can be readily incorporated in the existing aerobic theory to establish a comprehensive model that allows an accurate description of the single sludge nitrification-denitrification activated sludge process. Experimental data is presented to test the theory over a wide range of operational conditions (influent flow and load patterns, sludge ages, temperatures), reactor configurations (pre- and post-denitrification systems, completely mixed and plug flow reactors) and operational modes (continuously anoxic reactors, alternating anoxic and aerobic periods) in a reactor. Also, in Chapter 4, it will be shown that the empirical equations for nitrate removal (Eqs (1.3 and 1.4)) can be reformulated and that the empirical constants α, K_1, K_2 and K_3 can be interpreted in terms of the fractional composition of the influent COD and of the basic kinetic constants defining the utilization rates of easily and slowly biodegradable substrates. A convenient experimental method to determine the fractional composition of the influent COD and the relevant kinetic constants is presented.

The general model that was developed describes the behaviour of anoxic/aerobic activated sludge processes for specified values of the operational parameters (sludge age, volumes of anoxic and aerobic reactors, recycle flows). It does not, however, provide information on the optimal values of these operational parameters for maximum nitrogen removal. For this purpose an optimization procedure was developed which is described in Chapter 5. The basic considerations for optimization of nitrogen removal are:

(1) Nitrification must be efficient.

(2) Denitrification must be maximum and, if possible, complete.

Expressions are derived to calculate the optimal values for the operational parameters as a function of the relevant sewage characteristics and kinetic constants. The optimization procedure allows a rational selection of the operational parameters to be made for influents with any specified characteristics, to give maximum nitrate removal by nitrification and denitrification.

CHAPTER TWO

NITROGEN REMOVAL FROM WASTEWATER - A LITERATURE REVIEW

1. INTRODUCTION

Practical procedures of nitrogen removal from waste streams have received general attention only recently. However, a significant amount of research work has been carried out over the past two decades into the various methods and processes by means of which nitrogen can be removed. Different methods of nitrogen removal have been proposed and investigated. These can be divided into two groups: Non-biological and biological. Non-biological methods include chemical, (breakpoint chlorination; Pressly, Bishop, Pinto and Cassel, 1973) physical chemical (ammonia ion exchange; EPA, 1975) and physical processes (ammonia stripping with air; Slechta and Culp, 1976). These methods will not be discussed further.

The great majority of the research efforts has been concerned with biological nitrogen removal, by the sequential processes of nitrification and denitrification. There seems to be a general agreement that biologically mediated nitrogen removal is the most reliable and econonical of all the alternatives that are technically feasible. The two aspects of biologically mediated nitrogen removal - nitrification and denitrification - will be reviewed in Sections 2 and 3 respectively.

2. NITRIFICATION

2.1 Stoichiometry of nitrification

There is general agreement on the basic mechanism of the nitrification reaction. It is the biologically mediated oxidation of ammonia ions to nitrate ions in an aqueous environment. The nitrification reaction takes place in two steps: Ammonia is oxidized to nitrite and nitrite is oxidized to nitrate. Many types of bacteria (both autotrophic and heterotrophic) have been shown to be capable of mediating to some degree one or the other of the two oxidation steps

(Painter 1970) but nitrification in wastewater treatment is mediated almost exclusively by autotrophic bacteria belonging to the Nitrobacteraceae family; species of the Nitrosomonas generum mediate the oxidation of ammonia to nitrite and species of the Nitrobacter generum oxidize the nitrite ions to nitrate ions.

Autotrophic nitrifying organisms derive all their energy requirements for cellular metabolism (i.e. growth and maintenance) from the free energy released by the oxidation of the inorganic substrates ammonia and nitrite. The bacteria are specific in their energy source: Nitrosomonas can use only ammonia and Nitrobacter only nitrite as energy source. The redox reactions of oxidation of ammonia and nitrite respectively may be expressed as:

$$NH_4^+ + 1{,}5\ O_2 \rightarrow NO_2^- + H_2O + 2H^+ \qquad (2.1)$$

and

$$NO_2^- + 0{,}5\ O_2 \rightarrow NO_3^- \qquad (2.2)$$

The free energies released by the oxidations have been estimated at about 69 kcal mol^{-1} for ammonia to nitrite and about 18 kcal mol^{-1} for nitrite to nitrate respectively (EPA, 1975). Part of the energy released on oxidation of ammonia to nitrite is used for synthesis of Nitrosomonas (growth) during which some of the ammoniacal nitrogen is assimilated into bacterial protoplasm, using CO_2 as the carbon source; the remainder is lost as heat. Haug and McCarty (1971) accepted a structural formula of $C_5H_7O_7N$ for Nitrosomonas protoplasm and assumed the following synthesis reaction :

$$5CO_2 + NH_4^+ + 2H_2O \rightarrow C_5H_7NO_2 + 5O_2 + H^+. \qquad (2.3)$$

When growth of Nitrosomonas occurs, ammonia is used both as an energy source (Eq 2.1) and as a material source (Eq 2.3). The proportion of ammonia oxidized to ammonia utilized can be calculated from the yield coefficient for Nitrosomonas. (The increase in mass of Nitrosomonas per unit mass of ammonia utilized.) Table 2.1 shows reported values for the yield coefficient for Nitrosomonas. Accepting the maximum reported value, Y_n = 0,15 mg Nitrosomonas.mg NH_4^+-N^{-1}, a reaction equation incorporating both oxidation of ammonia to

Table 2.1 Experimental cell growth yield coefficients for Nitrosomonas and Nitrobacter.

NITROSOMONAS		NITROBACTER	
Y_n	Reference	Y_{ni}	Reference
0,06	Baas-Becking and Paules (1927)	0,00	Marais and Ekama (1976)
0,05	Downing, Painter and Knowles (1964)	0,05	Lees and Simpson (1957)
0,05	Downing, Tomlinson and Truesdale (1964)	0,04 to 0,07	Gould and Lees (1960)
0,05	Knowles, Downing and Barrett (1965)	0,02	Boon and Landelot (1962)
0,03	Loveless and Painter (1968)	0,02	Downing, Painter and Knowles (1964)
0,13	Skinner and Walker (1961)	0,02	Haug and McCarty (1971)
0,15	Sutton, Yank, Monaghan and Murphy (1979)		
0,15	Haug and McCarty (1971)		

nitrite and maximum synthesis of Nitrosomonas can be derived by balancing Eqs (2.1 and 2.3):

$$55NH_4^+ + 5CO_2 + 76O_2 \rightarrow C_5H_7O_2N + 54NO_2^- + 52H_7O + 109H^+ \qquad (2.4)$$

From Eq (2.4) 55 moles of NH_4^+ utilized generate 54 moles NO_2^-; 1 mole is incorporated in the biological mass of Nitrosomonas (a check on Eq (2.4) can be made by calculating the yield coefficient from it: 55 moles NH_4^+ = 770 g N produce 1 mole $C_5H_7O_2N$ = 113 g Nitrosomonas i.e. Y_n = 0,15 mg Nitrosomonas.mg NH_4^+-N^{-1}). Thus even for the assumed maximum value of the yield coefficient a fraction of 54/55 = 98 per cent of the utilized ammonia is oxidized and only 2 per cent is synthesized. In actuality the ratio (ammonia oxidized/ammonia utilized) may even be closer to unity because (1) the yield coefficient is likely to be smaller than 0,15 mg Nitrosomonas.mg NH_4^+-N^{-1} (see

Table 2.1) and (2) due to death of Nitrosomonas a certain fraction of the synthesized ammonia can possibly be reutilized.

Accepting Eq (2.4) as the reaction equation for maximum growth and Eq (2.1) the reaction equation for minimum growth of Nitrosomonas (i.e. no growth at all) the minimum and maximum oxygen requirement for utilization of ammonia can be calculated. From Eq (2.1) the maximum oxygen requirement is 1,5 moles O_2 per mole NH_4^+ utilized i.e. 3,43 mg O.mg NH_4^+-N^{-1}. Taking maximum synthesis into account one has (Eq 2.4) 76 moles O_2 for 55 moles NH_4^+ or 3,16 mg O.mg NH_3-N^{-1}.

From Eqs (2.1 or 2.4) nitrification affects the alkalinity. From Eq (2.1) two protons are released in the oxidation of ammonia to nitrite, i.e. the decrease of alkalinity is calculated as 2 moles per mole NH_4^+ oxidized or 7,14 mg $CaCO_3$.mg NH_3-N^{-1} oxidized. From Eq (2.4) the alkalinity effect is 109 moles H^+ per 55 moles of NH_4^+ or 7,08 mg $CaCO_3$.mg NH_3-N^{-1}.

In table 2.2 are listed the values of the oxygen requirement for oxidation of ammonia to nitrite and the associated alkalinity effect for synthesis of Nitrosomonas (i.e. calculated on the basis of Eq(2.4)), and, when considering only oxidation and no synthesis (i.e. calculated on the basis of Eq (2.1)). These conditions give the extreme values, i.e. the maximum and minimum for oxygen requirement and alkalinity decrease due to oxidation of ammonia to nitrite respectively.

Considering Nitrobacter and assuming the same structural formula as for Nitrosomonas a similar analysis as above gives a reaction equation for oxidation of nitrite and synthesis of Nitrobacter. From Table 2.1 the yield coefficient for Nitrobacter is probably less than the maximum reported value, Y_{ni} = 0,07 mg Nitrobacter.mg NO_2-N^{-1}. Taking this value as a maximum a reaction equation incorporating both oxidation of nitrite to nitrate and synthesis of Nitrobacter is obtained by balancing Eqs (2.2 and 2.3):

$$115\ NO_2^- + 52{,}5\ O_2 + 5CO_2 + NH_4^+ + 2H_2O \rightarrow C_5H_7O_2N + H^+ + 115\ NO_3^- \qquad (2.5)$$

It is now possible to calculate the maximum and minimum values for oxygen requirement for oxidation of nitrite to nitrate and

Table 2.2 Maxima and minima of oxygen requirement and alkalinity effect for oxidation of ammonia to nitrite, of nitrate to nitrite and of ammonia to nitrate. (Maximum values of Y_n = 0,15 mg VSS. mg NH_4^+-N^{-1} and Y_{ni} = 0,07 mg VSS.mg NO_2-N are assumed).

Reaction	O requirement (mg O.mg N^{-1})		Alkalinity effect (mg CaCo. mg N^{-1})	
	Max	Min	Max	Min
$NH_4^+ \rightarrow NO_2^-$	3,43	3,16	7,14	7,08
$NO_2^- \rightarrow NO_3^-$	1,14	1,11	0,0	0,0
$NH_4^+ \rightarrow NO_3^-$	4,57	4,27	7,14	7,08

the associated alkalinity change respectively. In Table 2.2 the maximum and minimum values for the oxygen requirement for oxidation of nitrite to nitrate and the associated alkalinity effect respectively are listed. Also indicated are the theoretical maxima and minima for the overall process of oxidation of ammonia to nitrate by the joint effect of the two organisms.

In Table 2.3 experimental values of the oxygen requirement for nitrification are listed. It may be noted that the reported values tend to be equal or near to the theoretical maximum or minimum values. However the difference between these extremes is small.

In Table 2.4 reported experimental values of alkalinity decrease due to nitrification are shown. The observed alkalinity changes in wastewater treatment plants in general are significantly smaller than the minimum of 7,08 mg $CaCO_3$.mg NH_3N^{-1} (Table 2.2). Scearce, Benninger, Weber and Sherrard (1980) found the explanation for this phenomenon: in domestic wastewaters a significant fraction of the

Table 2.3 *Experimental oxygen requirements for oxidation of ammonia to nitrate.*

O_{n1} *	O_{n2} **	O_{nt} ***	Reference
3,43	1,14	4,57	Ekama and Marais (1978)
3,43	1,14	4,57	Bonazzi (1923)
3,22	1,11	4,33	Montgomery and Borne (1966)
3,22	1,11	4,33	Wezernak (1967)

* O_n = oxygen required for oxidation of ammonia to nitrite ($mgO_2.mgN^{-1}$)

** O_n = oxygen required for oxidation of nitrite to nitrate ($mgO_2.mgN^{-1}$)

*** O_{nt} = oxygen required for oxidation of ammonia to nitrate ($mgO_2.mgN^{-1}$)

Table 2.4 *Experimental values showing decrease of alkalinity (as mg $CaCo_3$) per mg of nitrified TKN.*

mg $CaCO_3$.mg N^{-1}	Reference
6,3 - 7,4	Osborn (1965)
7,3	Haug and McCarty (1971)
6,4	Mulbarger (1971)
6,0	Horskotte, Niles, Parker and Caldwell (1974)
7,1	Newton and Wilson (1973)
5,4 - 7,2	Benninger and Sherrard (1978)
6,2 - 6,5	Bishop and Farmer (1978)
6,5	Gasser, Chen and Miele (1978)

influent TKN usually is present as organic nitrogen ($R\text{-}NH_2$) and must be hydrolyzed (ammonified) before Nitrosomonas can utilize it as a substrate. The ammonification reaction can be written as:

$$R\text{-}NH_2 + H_2O + H^+ \rightarrow R\text{-}OH + NH_4^+ \qquad (2.6)$$

From Eq (2.6) the ammonification reaction consumes hydrogen ions, i.e. alkalinity is produced. Stoichiometrically the increase in alkalinity is 3,57 mg $CaCO_3$.mg N^{-1} hydrolyzed. If ammonification is taken into account, Scearce *et al* (1980) showed that good correspondence between theoretical and experimentally observed alkalinity changes during nitrification in activated sludge plants is obtained.

From the above discussion it is apparent that the utilization of ammonia and nitrate by Nitrosomonas and Nitrobacter respectively is very closely approximated by considering only the oxidation steps (Eqs 2.1 and 2.2) i.e. the fraction of ammonia used for synthesis of the autotrophic cells is negligible compared to the fraction of ammonia or nitrite that is oxidized to affect this synthesis.

2.2 Kinetics of Nitrification

There appears to be general agreement in the literature that the growth of Nitrifiers can be described satisfactorily by the Monod equation (Monod, 1950):

$$(d\ X_n/dt)g = \mu_{nT} X_n \qquad (2.7)$$

and

$$\mu_{nT} = \mu_{nmT} \cdot N/(N+K_{nT}) \qquad (2.8)$$

where

X_n = concentration of Nitrosomonas or Nitrobacter (mg VSS. $^{-1}$)

μ_{nT} = specific growth rate of nitrifiers (mg VSS.mg VSS^{-1} d^{-1})

μ_{nmT} = maximum specific growth rate of nitrifiers (mg VSS.mg $VSS^{-1}.d^{-1}$)

K_{nT} = half saturation value (mg $N.\ell^{-1}$)

N = substrate concentration (mg $N.\ell^{-1}$ of ammonia or nitrite)

Index g refers to growth of nitrifiers.

Knowles, Downing and Barrett (1965) demonstrated experimentally that the first oxidation step, i.e. the oxidation of the ammonia ion to the nitrite ion by Nitrosomonas is the rate limiting step. In wastewater treatment nitrite is supplied solely by the oxidation of ammonia. Consequently the first oxidation step controls the nitrification process. The second oxidation step is so rapid that the intermediate product, nitrite, normally is not encountered in any significant concentration in nitrifying wastewater treatment plants. For this reason, in considering the kinetics of nitrification, only the characteristics of Nitrosomonas are of consequence.

Concomitant with a rate of growth of Nitrosomonas there is an associated rate of death. Equation (2.7) can be adapted readily to include the death rate, $(d X_n/dt)_d$, of Nitrosomonas. The nett growth rate, $(d X_n/dt)_n$, due to synthesis and the death rate is given by:

$$(d X_n/dt)_n = (d X_n/dt)_g + (d X_n/dt)_d = (\mu_{nT} - b_{nT}) X_n \qquad (2.9)$$

where

b_{nT} = specific death rate constant of Nitrosomonas (mg VSS.mg $VSS^{-1}.d^{-1}$)

Many research workers have reported experimentally observed values of the kinetic constants μ_{nmT}, b_{nT} and K_{nT}. Tables (2.5, 2.6 and 2.7) list reported experimental values of these constants using activated sludge systems for nitrification and sewage as a source for the ammonia substrate. The values of the constants show wide variation, the possible causes for this variation will be discussed in the next section. For such a discussion to be of practical use it is necessary first to discuss in general the effect of each of the constants on the process behaviour.

Table 2.5 *Reported values for the maximum specific growth rate of* *Nitrosomonas* *in activated sludge systems.*

μ_{nmT} (d^{-1})	T (°C)	μ_{nm20} * (d^{-1})	Reference
0,17	25	0,10	Melamed, Saliternik and Wachs (1970)
0,17	20	0,17	Wilson and Marais (1976)
0,33	20	0,33	Downing, Painter and Knowles (1964)
0,33-0,65	20	0,33-0,65	Ekama and Marais (1978)
0,34-0,40	12	0,86-1,01	Gujer and Jenkins (1974)
0,37	23	0,26	Balakrishnan and Eckenfelder (1969)
0,4 - 0,5	14	0,8 - 1,0	Gujer (1977)
0,5	20	0,5	Lawrence and Brown (1973)
0,53	25	0,26	Sutton *et al* (1979)
0,57	16	0,91	Wuhrmann (1965)
0,65-0,85	21	0,58-0,76	Gujer and Jenkins (1974)
0,71	20	0,71	Loehr *et al* (1973)
0,94	29	0,33	Lijklema (1973)
1,08-1,44	23	0,76-1,02	Poduska and Andrews (1974)

* The data was obtained at different temperatures.

For comparison a reference value at 20°C has been calculated based on the temperature dependency relation suggested by Downing *et al* (1964) :

$$\mu_{nmT} = \mu_{nm20} (1,123)^{T-20}$$

Table 2.6 *Reported values for the death rate constant of* ***Nitrosomonas*** *in activated sludge systems.*

b_{nT} (d^-)	T (°C)	b_{n20} * (d^-)	Reference
0,0	20	0,0	Downing Painter and Knowles (1964)
0,0	15	0,0	Downing Painter and Knowles (1964)
0,0	10	0,0	Gujer (1979)
0,04	20	0,04	Ekama and Marais (1978)
0,071	20	0,07	Lijklema (1973)
0,12	29	0,09	Lijklema (1973)
0,12	23	0,11	Poduska and Andrews (1974)

* The data was obtained at different temperatures.

For comparison a reference value at 20°C has been calculated based on the temperature dependency relation suggested by Dold, Ekama and Marais (1980) :

$$b_{nT} = b_{n20} (1,029)^{T-20}$$

Table 2.7 *Reported values for the half saturation concentration of ammonia, K_n, in the Monod equation for growth of Nitrosomonas in actived sludge systems.*

K_{nT} (mg N.ℓ^{-1})	T (°C)	K_{n20} * (mg N.ℓ^{-1})	Reference
0,06	23	0,04	Poduska and Andrews (1974)
0,2	15	0,1	Downing *et al* (1964)
0,2	20	0,2	Downing *et al* (1964)
0,2	10	0,64	Gujer (1977)
0,25 - 0,37	25	0,14 - 0,21	Melamed, Saliternik and Wachs (1970)
0,5	14	1,0	Ekama and Marais (1978)
1,0	20	1,0	Ekama and Marais (1978)
1,0	20	1,0	Lijklema (1973)
0,0	29	2,8	Lijklema (1973)

* The data was obtained at different temperatures.

For comparison a reference value has been calculated based on the temperature dependency relation suggested by Dold, Ekama and Marais (1980) :

$$K_{nT} = K_{n20} \ (1,123)^{T-20}$$

The influence of the kinetic constants μ_{nmT}, b_{nT} and K_{nT} on the kinetics of nitrification and the effluent ammonia concentration can be evaluated by considering a single reactor completely mixed activated sludge system under steady state conditions (Fig 2.1). Under steady state conditions there is a constant rate of abstraction and wastage of sludge (wastage is assumed to be from the reactor). With a waste flow q per day from a reactor with volume V the mass of daily wasted sludge is $q.X_n$, i.e. a fraction q/V of the total mass of sludge $V.X_n$ in the reactor. Hence the solids retention time, or sludge age, is given by $R_s = 1/(q/V)$ or $q = V/R_s$. A mass balance on X_n over the reactor gives:

$$V.(dX_n/dt) = V.(dX_n/dt)_g + V.(dX_n/dt)_d + V(dX_n/dt)_w. \qquad (2.10)$$

where indices g, d and w refer to growth, death and wastage of Nitrosomonas respectively. Under steady state conditions there is no change in the concentration of Nitrosomonas with time: $dX_n/dt = 0$. Inserting Eq (2.9), in Eq (2.10):

$$\mu_{nmT}X_n - b_{nT} - X_n/R_s = 0 \qquad (2.11)$$

Substituting Eq (2.8) in (2.11) and solving for N_a:

$$N_a = K_{nT}(b_{nT} + 1/R_s)/(\mu_{nmT} - (b_{nT} + 1/R_s)) \qquad (2.12)$$

From Eq (2.12) it is evident that the effluent ammonia concentration is influenced by each of the three kinetic constants of nitrification, μ_{nmT}, b_{nT} and K_{nT} and by the sludge age of the system. It is of importance to note that the effluent ammonia concentration does not depend on the influent TKN concentration.

From Eq (2.12) there is an increase in the effluent ammonia concentration, N_a, with decreasing sludge age. At the minimum sludge age for nitrification, R_{sm}, the effluent ammonia concentration is equal to the influent concentration available for nitrification, N_p. The value of the minimum sludge age for nitrification can be calculated by substituting N_p for N_a in Eq (2.12) and solving for R_{sm}:

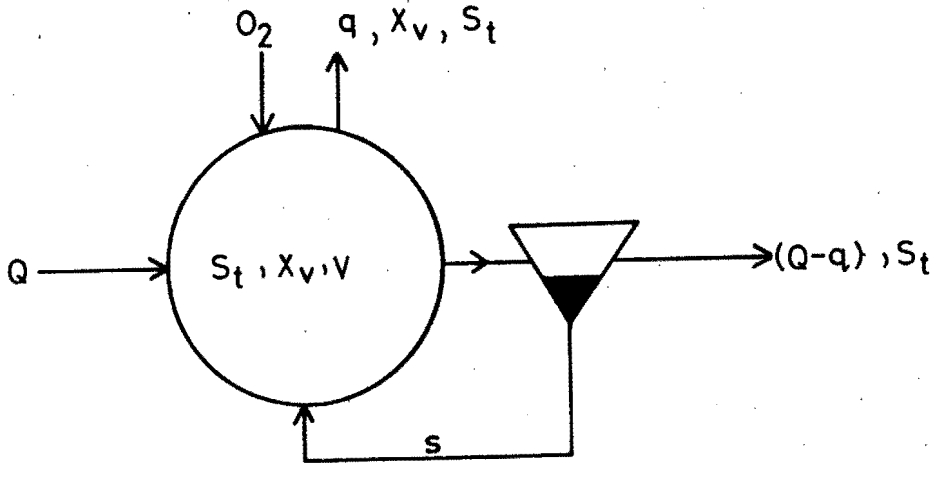

Fig 2.1:*Completely mixed activated sludge process under constant flow and load conditions.*

$$R_{sm} = (1 + K_{nT}/N_p) / (\mu_{nmT} - b_{nT}) \qquad (2.13)$$

For any $R_s > R_{sm}$ the effluent ammonia concentration can be plotted as a function of the sludge age for the different reported values of μ_{nmT}, b_{nT} and K_{nT}. Taking 14°C as a reference temperature and accepting as average experimental values $\mu_{nm14} = 0,25\ d^{-1}$, $K_{n14} = 0,5$ mg NH_3-N.ℓ^{-1} and $b_{n14} = 0,034\ d^{-1}$, the influence of the numerical values of these constants on the effluent ammonia concentration is shown in Figs 2.2(a, b and c).

Effect of the value of the maximum specific growth rate of Nitrosomonas, μ_{nmT}.

In Fig 2.2a the effluent ammonia concentration is shown plotted as a function of the sludge age for $\mu_{nm14} = 0,35$ and $\mu_{nm14} = 0,15\ d^{-1}$

with $b_{n14} = 0{,}034\ d^{-1}$ and $K_{n14} = 0{,}5$ mg NH_3.N.ℓ^{-1} for both. The two μ_{nm14} values represent a very high and low experimental value respectively. From Fig 2.2a the minimum sludge age for nitrification is strongly affected by the μ_{nm14} value. For sludge ages much greater than the minimum there is very little difference between the two effluent ammonia concentrations.

Effect of the value of the half saturation concentration, K_{nT}

In Fig 2.2b the effluent ammonia concentration is shown plotted as a function of sludge age for $K_{n14} = 1{,}0$ and $K_{n14} = 0{,}05$ mg NH_3-N.ℓ^{-1}, keeping $\mu_{nm14} = 0{,}25 d^{-1}$ and $b_{n14} = 0{,}034\ d^{-1}$ for both. Changes in the value of K_n do not affect the minimum sludge age of nitrification but when nitrification takes place the effluent ammonia concentration increases proportionally with K_n, i.e. the higher K_n the higher the effluent ammonia concentration. However, even if K_n is as high as 2 mg NH_3-N.ℓ^{-1} the effluent ammonia concentration will still be less than 1 mgN.ℓ^{-1} at long sludge ages. At $R_s > (1{,}25 \text{ to } 1{,}5)R_{sm}$, the effluent ammonia concentration stabilizes at a low value proportional to the K_n value so that for efficient nitrification the operating sludge age should be at least $(1{,}25 \text{ to } 1{,}5)R_{sm}$. For such a sludge age the effluent ammonia concentration will be low, irrespective of the K_n value. Hence the exact numerical value of K_n in general is not of practical significance.

Effect of the value of the death rate constant, b_{nT}

In Fig 2.2c the effluent ammonia concentration is shown plotted for $b_{n14} = 0{,}05$ and $b_{n14} = 0{,}0\ d^{-1}$, keeping $\mu_{nm14} = 0{,}25\ d^{-1}$ and $K_{n14} = 0{,}5$ mg NH_3-N.ℓ^{-1} for both. From Fig 2.2c the numerical value of b_{nT} affects both the minimum sludge age for nitrification and the effluent ammonia concentration but its effect on either is small compared to the influence of μ_{nmT} on the minimum sludge age and of K_{nT} on the effluent ammonia concentration respectively. Hence knowledge on the exact value of b_{nT} is of limited practical importance.

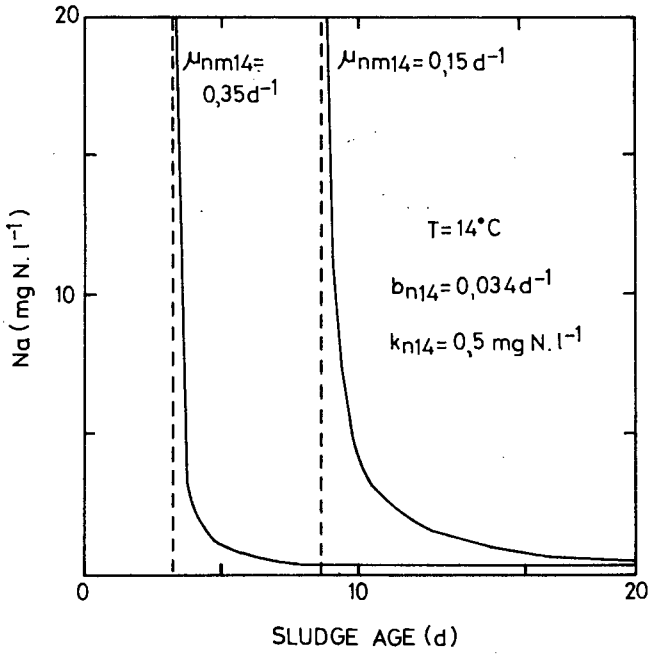

Fig.2.2a *Effluent ammonia concentration as a function of sludge age for maximum specific growth rates* $\mu_{nm14} = 0,15$ *and* $\mu_{nm14} = 0,35d^{-1}$, *keeping* $K_{nT} = 0,5$ *mgN*ℓ^{-1} *and* $b_{nT}=0,034d^{-1}$.

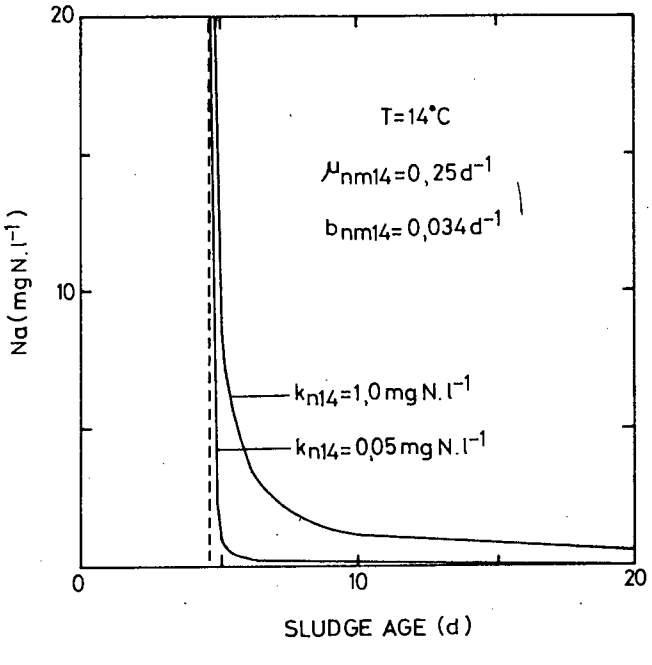

Fig 2.2b *Effluent ammonia concentration as a function of sludge age for half saturation values* $K_{n14} = 1,0$ *and* $K_{n14} = 0,05$ *mg N.*ℓ^{-1} *keeping* $\mu_{nm14} = 0,25^{-1}$ *and* $b_{n14} = 0,034d^{-1}$.

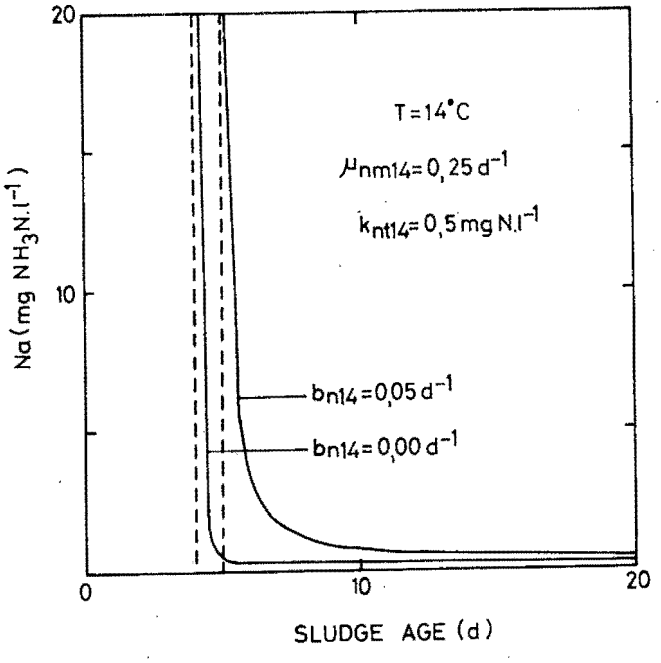

Fig 2.2c *Effluent ammonia concentration as a function of sludge age for death rate constants* $b_{n14} = 0,042$ *and* $b_{n14} = 0,0d^{-1}$ *keeping* $\mu_{nm14} = 0,25d^{-1}$ *and* $K_{n14} = 0,25\ mg\ N.\ell^{-1}$.

From an examination of Figs 2.2a, b and c, it can be concluded that the influence of the value of μ_{nmT} on the effluent concentration is far more pronounced than that of either of the other two constants b_{nT} or K_{nT}. The fact that so widely varying numerical values for b_{nT} and K_{nT} are reported (Tables 2.6 and 2.7) is not of practical significance. The efficiency of nitrification is determined principally by the maximum specific growth rate of the nitrifiers and the operating sludge age. Indeed,if the sludge age is about 25 per cent longer than the minimum sludge age, efficient nitrification will occur irrespective of the values of μ_{nmT}, b_{nT} and K_{nT}. Hence the important point of nitrification is to determine the minimum sludge age for nitrification, and this sludge age is set mainly by the maximum specific growth rate of nitrifiers, Eq (2.13). Therefore the variation of reported μ_{nmT} values (Table 2.5) has far more important practical consequences than the variation in the other two constants b_{nT} and K_{nT}.

2.3 Factors affecting μ_{nmT}, K_{nT} and b_{nT}

2.3.1 Presence of toxic and inhibitory substances

The maximum specific growth rate of Nitrosomonas, μ_{nm}, in pure cultures is about 1,2 mg $X_n.mgX_n^{-1}.d^{-1}$ at 20°C (Skinner and Walker, 1962, Loveless and Painter, 1968, Engel and Alexander, 1958). Reported values of μ_{nm} on nitrifying activated sludge systems treating municipal sewage are generally lower (see Table 2.5) ranging from about 0,25 to 1,0 d^{-1} at 20°C. The principal cause of these lower values appears to be the presence of industrial waste in the influent flow.

Ekama and Marais (1978) reported a μ_{nm20} value of 0,67 d^{-1} for waste water from the Strandfontein-Cape Town outfall, which contains a high fraction of domestic wastewater against a value of μ_{nm20} = 0,33 d^{-1} for sewage from the Daspoort-Pretoria outfall which contains a significant fraction of industrial discharge. An example of a waste flow containing a very high fraction of industrial wastes is that from the Athlone-Cape Town outfall. From an analysis of activated sludge nitrification data obtained by Wilson and Marais (1976) on the Athlone waste flow a μ_{nm20} of only 0,17 d^{-1} was calculated (see Chapter 4).

The data above show that the effect of industrial waste on μ_{nm20} is inhibitory rather than toxic. This is an important distinction for it implies that, in design, nitrification of an industrial waste flow can be assured by increasing the sludge age appropriately until it exceeds the minimum sludge age for nitrification associated with the μ_{nm} value of such a wastewater.

The dependence of the maximum specific growth rate of Nitrosomonas on the wastewater source implies that for all practical purposes, μ_{nm20} should be considered as a sewage characteristic, and that its value should be determined experimentally for a particular waste flow.

In general, nitrifiers are more susceptible to the effect of inhibitory substances than the heterotrophic bacteria in activated

sludge. A possible explanation for this is that, while the conversion of ammonia to nitrate depends upon the growth of two specific genera of bacteria (Nitrosomonas and Nitrobacter), the conversion of organic material to CO_2 can be performed by a variety of heterotrophic organisms. These heterotrophic organisms have different tolerances for the different substrates in sewage. If a particular organic substrate inhibits the growth of one group of heterotrophs, it is likely that another group may benefit from it, and a population shift will occur to the one that can use the organic substrate more effectively. As a consequence only a transitory reduction of the efficiency of the heterotrophs during a period of adaption to the substrate may be observed. In contrast, when Nitrosomonas are affected by inhibitory substances, this has an immediate and lasting effect on the maximum rate of nitrification as there are no other micro-organisms (at least not in sufficient quantity) that can substitute for Nitrosomonas.

In virtually all the research reported on the effect of inhibitory substances on nitrification is measured as a decrease in the rate of nitrification and linked directly to a proportional decrease in the maximum growth rate of nitrifiers. It is possible that the other kinetic constants of nitrification (particularly K_n) are also affected, but no information on this could be found in the literature.

2.3.2 Temperature sensitivity.

The growth rate of Nitrosomonas increases with increasing temperature, the maximum rate occurring at about 30 to 35°C. Temperatures higher than 35°C cause a decrease in the growth rate due to denaturation of enzymes of the biochemical pathways. The maximum specific growth rate, μ_{nmT}, in many cases has been found to double for approximately every 6°C increase in temperature.

It has become customary among research workers in the field of wastewater treatment to express the variation of the growth rates of the organisms in terms of the simplified Arrhenius equation. Being a simplified equation its range of applicability will be restricted; consequently, care should be taken in using this equation for extrapolating outside the range in which the constants were determined.

The simplifed Arrhenius equation is expressed as:

$$\mu_{nmT} = \mu_{nmt}\theta^{(T-t)} \qquad (2.15)$$

where

μ_{nm} = maximum specific growth rate ((d^{-1})

T = operational temperature (^{o}C)

t = reference temperature (^{o}C)

θ = temperature dependency constant

The following values of μ_{nmT} and θ have been reported in the literature:

Downing *et al* (1964) reported the following equation from experimental data on an activated sludge plant treating waste with a high industrial faction

$$\mu_{nmT} = 0,33\ (1,123)^{(T-20)} \qquad (2.16)$$

Knowles *et al* (1965) reported that the variation of maximum specific growth rate of Nitrosomonas with temperature in samples from the Thames River estuary is given by

$$\log_{10}\mu_{nmT} = 0,0413\ T - 0,944 \qquad (2.17)$$

which in the Arrhenius form, with $20^{o}C$ as base, is equivalent to

$$\mu_{nmT} = 0,762\ (1,100)^{(T-20)} \qquad (2.18)$$

From data reported by Buswell, Shiota, Lawrence and Van Meter (1954) the variation of the specific growth rate of Nitrosomonas in a pure culture was found to be about 8,2 per cent per ^{o}C, which is equivalent to a θ value of 1,082.

Gujer (1977), from pilot scale activated sludge process studies, found excellent correlation with experimental data

for short term (daily) and long term (seasonal) temperature variations, both of which follow the Arrhenius formulation. For the long term effects Gujer presents the following equation, applicable between 6 and 14°C:

$$\mu_{nmT} = 0,29\ e^{0,11(T-10)} \qquad (2.18a)$$

which is equivalent to

$$\mu_{nmT} = 0,87\ (1,116)^{T-20} \qquad (2.19)$$

This μ_{nm20} value appears to be quite high but the θ value is very near to that reported by Downing *et al* (1964).

Lijklema (1973) found good agreement between theoretical predictions and experimental data obtained on full scale operations reported in the literature using a value of $\theta = 1,13$.

Ekama and Marais, in calibrating a model to simulate nitrification behaviour in activated sludge units operated by them found that for a temperature increase from 14°C to 20°C it was necessary to double the value of the maximum specific growth rate, which is equivalent to $\theta = 1,123$. This θ value was found to be applicable for different sewages with different μ_{nm20} values.

A summary of temperature dependency constants for the maximum specific growth rate of Nitrosomonas is presented in Table 2.8

Table 2.8 *Temperature dependency constants for the maximum specific growth rate of Nitrosomonas in activated sludge systems.*

θ	Temperature Interval (^{o}C)	Reference
1,116	19 - 21	Gujer (1977)
1,123	15 - 20	Downing *et al* (1964)
1,123	14 - 20	Ekama and Marais (1978)
1,130	20 - 30	Lijklema (1973)

From Table 2.8 there is good agreement on the effect of temperature on the maximum specific growth rate of Nitrosomonas between different investigators: The temperature dependency is severe; the μ_{nmT} value is reduced to half its initial values for a decrease of 6 to 7^{o}C in temperature.

With regard to the half saturation constant, K_n, in the Monod relationship for Nitrosomonas this kinetic constant has been found to increase with increasing temperature.

Knowles *et al* (1965) presented the following relationship:

$\log_{10} K_{nT} = 0,051\ T - 1,158$

for ammonia oxidation by Nitrosomonas. This equation is equivalent to:

$$K_{nT} = 0,73\ (1,125)^{(T-20)} \qquad (2.20)$$

Downing (1964) reported a half saturation value so low (0,18 mg $N.l^{-1}$) that the influence of temperature was not measurable.

In theoretical simulations of experimental full scale observation Lijklema (1973) used a value of K_n = 1,0 mg $N.l^{-1}$ at 20^{o}C and 8,0 at 29^{o}C.

A summary of the different values of the saturation constant for Nitrosomonas at various temperatures is given in Table 2.9. The θ_{ns} values between reported data show little agreement. However because K_n is small and its value does not affect the nitrification process greatly, error in the temperature dependency is not of significance. Insofar as the temperature dependency of the death rate constant of Nitrosomonas, b_{nT}, is concerned, the only information available seems to be that provided by Marais and Ekama (1978). They assumed that the temperature dependency of the death rate constant for Nitrosomonas, θ, is the same as that for heterotrophic bacteria*. They proposed the following temperature dependency relationship:

$$b_{nT} = b_{n20}(1,029)^{T-20} \qquad (2.21)$$

This temperature dependency can be considered as only approximate but from a large number of simulations of nitrification behaviour by means of the general mathematical model of Dold, Ekama and Marais (1980), it was concluded that the use of Eq (2.21) resulted in good agreement between experimental and simulated nitrification responses.

2.3.3 pH Sensitivity

The activity of nitrifying organisms is very sensitive to the hydrogen ion concentration of the mixed liquor. Optimal nitrification rates occur between values of 7,0 and 8,5 with the growth rates falling rapidly below pH 6,5 and above 9,0. However, various growth rate-pH relationships are reported in the literature:

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* The death rate constant for heterotrophs was determined experimentally by Marais and Ekama (1976) (see Chapter 3) and its temperature dependency, also determined from analysis of experimental data, as $b_{hT} = 0,24(1,029)^{T-20}$

Table 2.9 *Temperature dependency coefficients of the half saturation concentration of ammonia in the oxidation by Nitrosomonas in activated sludge systems*

θ	Temperature °C	Reference
1,0	15 - 20	Downing *et al* (1964)
1,123	14 - 20	Ekama and Marais (1978)
1,175	10 - 20	Gujer (1977)
1,26	20 - 30	Lijklema (1973)

Hoffman and Lees (1953), utilizing pure cultures, presented a curve with a maximum rate at pH = 8,4 and with rates of about half the maximum at pH = 7,0 and pH = 9,0 respectively.

Engel and Alexander (1958), also utilizing pure cultures, found that the maximum activity for Nitrosomonas occurred between 7,0 and 9,0, the activity dropping steeply outside this range. Loveless and Painter (1968), in an attempt to explain these differences, demonstrated that some types of metal ions adversely affected the growth rate of Nitrosomonas. They found that when using a medium composed of distilled water contaminated with 0,05 to 0,08 mg/ℓ of copper and 0,05 to 0,10 mg/ℓ of zinc, the growth rate versus pH curve had a definite peak at pH 8,0. When the heavy metal contaminants were excluded from the distilled water medium the curve obtained was virtually flat in the range pH 7,0 to 8,2, with the growth rate in this range greater than the previously obtained peak.

Downing *et al* (1964) found that in activated sludge systems the rate of nitrification was approximately constant in the range $7,2 < pH < 8,0$ but decreases rapidly for pH = 7,2

and becomes negligibly small for pH = 6,0. He suggested a linear decrease of μ_{nm} from a maximum at pH = 7,2 to $\mu_{nm} = 0$ at pH = 6,0.

An investigation into the change of the rate of nitrification in activated sludge units operated at 23°C for different mixed liquor pH values was undertaken by Malan and Gouws (1966). Their results, given in Table 2.10, show a quite marked decrease of the rate of nitrification, when the pH decreased from 8,0 to 7,0.

A major problem in investigations into the rate of nitrification in activated sludge processes and the effect of pH on it is that the rate depends upon the sewage characteristics and can vary considerably from one batch to another. Furthermore, because of the characteristics of the Monod equation, steady state investigations into the kinetics of nitrification (and the values of the relevant constants) are virtually worthless (Ekama and Marais, 1978). In most investigations into the effect of pH, the nitrification rates *per se* were measured. There appears to be no quantitative investigation into the effect of pH on each of the kinetic constant μ_{nm}, K_n and b_n respectively. It would appear that changes in the nitrification rate with pH are lumped onto changes in μ_{nm}, probably due to lack of data on changes of b_n and K_n with pH. Ekama and Marais (1978) accepted this approach and suggested the following expression to incorporate the effect of pH on nitrification:

$$\mu_{nmp} = \mu_{nm} \phi^{(pH-7,2)} \qquad (2.22)$$

where

μ_{nmp} = maximum specific growth rate of Nitrosomonas at a particular pH $\leq$ 8,5

μ_{nm} = maximum specific growth rate of Nitrosomonas at pH 7,2

ϕ = 1,0 for 7,2 $\leq$ pH $\leq$ 8,5

ϕ = 2,35 for pH $\leq$ 7,2

Table 2.10 *Variation of nitrification rate at different mixed liquor pH values (after Malan and Gouws, 1966)*

Mixed liquor pH	5,0	6,5	7,2	8,0
Nitrification rate	0,0	0,14	0,44	0,62

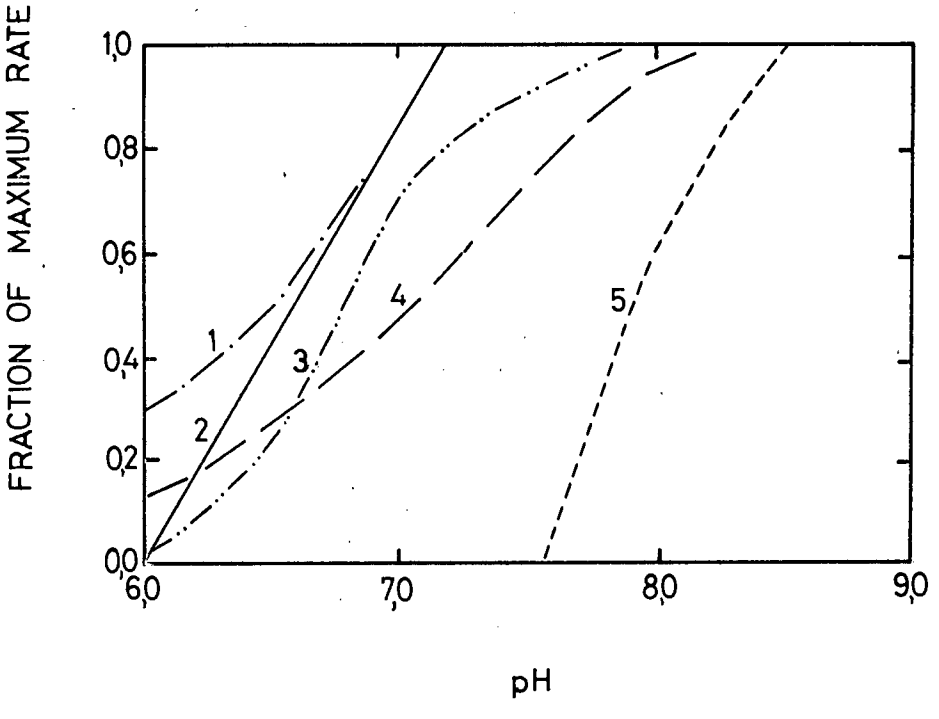

Fig.2.3 *Maximum nitrification rate as a function of mixed liquor pH (after EPA, 1975).*

(1) : Ekama and Marais (1978).

(2) : Downing et al (1966).

(3) : Malan and Gouws (1966).

(4) : Sawyer et al (1973).

(5) : Meyerhof (1917) (Pure culture).

Ekama and Marais obtained their relationship by calibration of their model from simulations of the activated sludge process against experimental data, using waste flows of widely varying characteristics. Using Eq (2.22) they obtained good agreement between theoretical and experimental values. The values obtained from Eq (2.22) agree approximately with those suggested by Downing *et al* (1964).

Some nitrification rate - pH relationships are shown in Fig 2.3. The relationship described by the model proposed by Ekama and Marais (1978) is also indicated.

2.3.4 Dissolved oxygen sensitivity

High dissolved oxygen concentrations (up to 33 mg $0.\ell^{-1}$) do not appear to affect nitrification rates significantly. Under low concentrations however, oxygen becomes a limiting nutrient, along with ammonia. It has been suggested that the influence of the concentration of dissolved oxygen be expressed as a Monod function (Stentrom and Poduska, 1980).

$$\mu_n = \mu_{nm} N_a/(N_a+K_n).DO/(DO+K_o) \qquad (2.21)$$

where

DO = concentration of dissolved oxygen.

K_o = half saturation value

The value attributed to the constant K_o by different research workers ranges from 0,3 to 2,0 mg $0.\ell^{-1}$. This wide range probably arises because the concentrations of DO in the bulk of the mixed liquor is not necessarily the same as that inside the biological floc where oxygen consumption takes place. The minimum concentration of dissolved oxygen that does not affect noticeably the rate of nitrification depends *inter alia* on the efficiency of dissolved oxygen transport from the bulk of the mixed liquor to the floc, and of the rate of transport inside the floc. Furthermore, in a large reactor the DO concentration will vary over the reactor volume due to the impossibility of ensuring instantaneous and complete mixing.

Hence factors such as mixing intensity in the aeration reactor and size of the flocs are likely to influence the minimum concentration of dissolved oxygen. For these reasons it is not really feasible to establish a generally applicable value of the minimum concentration. In practice every reactor will tend to have its own minimum DO concentration which may even vary with time. A popular assumption as to the minimum concentration of DO is that this value should be about 2 mg $0.\ell^{-1}$ at the surface of a reactor in order to ensure nitrification throughout the reactor.

In single sludge nitrification-denitrification systems it is important to establish (by observing the process response) the minimum DO concentration that does not affect the nitrification rate appreciably, because any oxygen that is transferred from an aerobic to an anoxic zone may tend to cause a decrease of the rate of denitrification in this reactor.

2.3.5 Effect of anoxic conditions

On single sludge nitrification-denitrification systems the sludge is placed alternately or sequentially in anoxic and aerobic environments. Anoxic/aerobic conditions have two effects on nitrification.

1. When the sludge is in an anoxic environment no nitrification takes place; all the nitrification must take place when the sludge is in an aerobic environment. Consequently the minimum sludge age for nitrification in an anoxic/aerobic system will be higher than in a comparable completely aerobic system.

2. As the nitrifiers cannot utilize their substrates in an anoxic environment to obtain energy,it is possible that an anoxic period has an inhibitory effect on nitrification in a subsequent aerobic period. This inhibitory effect can be either of a transitory or a permanent nature. Little information is available on the effect anoxic conditions, imposed for a limited period, have on the rate of nitrification in a subsequent aerobic period. Usually in the published literature efficient nitrification in anoxic/aerobic

activated sludge systems has been reported, but this does not imply the kinetic constants for nitrification in the aerobic environment of anoxic/aerobic systems are the same as in a comparable completely aerobic system: From Figs 2.2a, b & c, it is evident that if the sludge age is considerably higher than the minimum sludge age for nitrification, nitrification will be efficient irrespective of the numerical values of the kinetic constants for nitrification. Hence in systems with a series anoxic and aerobic reactor system, operating under constant flow and load conditions, it is virtually impossible to determine if the presence of the anoxic reactor affects the kinetic constants for nitrification in the aerobic reactor. Information on the kinetic constants can be obtained only under variable flow and load conditions i.e. when a situation is created in which nitrification is not efficient at all times. Then during a period of a high ammonia concentration in the aerobic reactor the *maximum* rate of nitrification can be determined and compared with the maximum rate in a comparable aerobic system. However, a transitory inhibitory effect of an anoxic period on nitrification in a subsequent aerobic period may not be picked up by such a procedure. At any time the mass of sludge leaving the anoxic reactor and entering the aerobic reactor is small compared to the mass of sludge in the aerobic environment and tends to mask the effects of the anoxic zone. Hence the nitrification response in the aerobic reactor of an anoxic/aerobic system will be very similar to that of a comparable aerobic system unless the period of transition from the inhibited state just after leaving the anoxic zone to the aerobic state is of the same order of magnitude as the retention time in the aerobic reactor.

From the discussion above it would appear that the most suitable experiment for the determination of inhibitory effects of an anoxic environment on the kinetic constants of nitrification is the

following: The entire mass of sludge in a system is placed in an anoxic environment for a certain period and then aeration is started. By observing the nitrification rate as a function of time, information about the influence of the anoxic period on nitrification in the subsequent aerobic period can be obtained. A survey of the literature indicates one instance where the above experiment was carried out: Downing *et al* (1964) reported that when a nitrifying sludge was submitted to an anoxic environment for four hours and aerated subsequently, the initial nitrification rate was zero but the original nitrification rate was re-established within a period of twenty minutes.

For rational design of single sludge anoxic/aerobic systems it is necessary to establish unequivocally the influence of anoxic periods on nitrification behaviour in subsequent aerobic periods. An experimental procedure to determine this influence, based on the approach discussed above, will be developed in Chapter 4.

2.4 Nitrification Models

Downing *et al* (1964) were the first to formulate a mathematical model for nitrification in the activated sludge system. They accepted that (1) The Monod equation expresses the growth rate of nitrifying bacteria , Nitrosomonas and Nitrobacter and hence the associated rate of oxidation to NO_2 and NO_3. (2) The oxidation of ammonia to nitrite is the rate limiting step, the oxidation of nitrite to nitrate is a relatively fast reaction. They established that a necessary condition for nitrification to occur continuously is that the rate of abstraction of Nitrosomonas through wastage must be less than or equal to the nett growth rate of these organisms in the system. This led to the important concept of a minimum sludge age of nitrification, discussed earlier in this Chapter (see Section 2.2.2). It was shown earlier that as the sludge age increases above the minimum for nitrification the nitrification efficiency increases very rapidly and nitrification is virtually complete for sludge ages $R_s > 1{,}25\ R_{sm}$. This is true irrespective of the values of u_{nmT}, K_{nT} and b_{nT}. This is the principle reason why one tends to observe either no nitrification or complete nitrification in a plant - it is unlikely that one should just strike the small range of sludge ages where nitrification is inefficient.

Ekama and Marais (1978) contributed to the development of kinetics of nitrification in that they made a distinction between ammonia and organic nitrogen in the influent TKN. They concluded that only ammonia can be utilized in the nitrification process. They established that organic nitrogen is converted to ammonia by active heterotrophic sludge. However, in the activated sludge process this conversion is normally incomplete and in nitrifying activated sludge plants the effluent TKN is composed principally of organic nitrogen.

For single reactor systems under constant flow and load conditions, if the kinetic constants for nitrification are known, an analytical solution for the effluent ammonia concentration may be found for a completely mixed reactor (as outlined in Section 2.2.2) and for a plug flow reactor (the latter by using the approach developed by Lawrence and McCarty (1970)). Due to the fact that nitrification in single reactor systems under constant flow and load conditions is virtually complete, the inverse, i.e. calculation of the kinetic constants for nitrification from the effluent ammonia concentration, is in general not possible.

For series reactor systems and/or variable flow and load conditions an analytical solution is not possible but the nitrification process can be simulated with a computer and the concentrations of ammonia, nitrate and Nitrosomonas calculated as a function of time and/or space. Several dynamic computer models have been proposed (Andrews and Poduska, 1974, Gujer, 1977, Lijklema, 1973); a particularly versatile model was developed by Ekama and Marais (1978) and later refined by Dold, Ekama and Marais (1980).

All these models use the Monod equation as the basic expression for nitrification and, in principle, all these models can be used to determine the kinetic constants of nitrification, μ_{nm}, K_n and b_n by calibration against experimental data. For this it is essential that the influent load is unsteady or that a series of completely mixed or a plug flow reactor is used, which allows the imposition of a transient state. When an unsteady load is used, nitrification will not be complete under peak load conditions. In the case of a series of completely mixed or plug flow reactor, nitrification near the point of introduction of the feed will be incomplete. Experimental data

obtained with these systems can be used for calibration of the theoretical models. However,it will be shown in Chapter 4 that these methods for determining the kinetic constants in general tend to affect adversely the experimental reproduceability of the process due to hydraulic effects related to the input flow (in the case of unsteady state), or flow in the reactor system (in the case of series reactors or a plug flow reactor). In the present work an experimental method to determine the kinetic constants for nitrification will be discussed in which hydraulic effects are minimized. This involves the use of a single completely mixed reactor under constant flow and load conditions with the reactor placed alternately in anoxic and aerobic state.

2.4.1 Effect of influent flow and load patterns

The authors who presented dynamic models for nitrification in the activated sludge process (Poduska and Andrews, 1974; Lijklema, 1973; Gujer and Jenkins, 1974; Ekama and Marais, 1978) have also reported experimental data on nitrification under varying flow and load conditions. They all agree that as the severity of variations of the influent TKN and/or the influent flow increases, so the efficiency of nitrification is reduced correspondingly. The apparent loss in efficiency can be shown to be the result of the kinetic response of a process when controlled by a Monod type of equation. The reason is that during periods of high load on the system (either by an increase of the influent TKN or the influent flow) the contact time in the reactor may be insufficient to oxidize the entire incoming load, and some ammonia will be discharged with the effluent. This in turn reduces the average mass of nitrifying organisms compared to a process under constant flow and load conditions, thus further reducing the nitrification rate and efficiency. Ekama and Marais have shown that the reduction in the efficiency of nitrification is particularly severe as $R_s \simeq R_{sm}$; at R_s longer than about 1,5 R_{sm} the reduction is negligible. This illustrates the importance of operating plants with sludge ages well in excess of the minimum sludge age for nitrification.

3. DENITRIFICATION

3.1 Stoichiometry of denitrification

Denitrification in its widest sense refers to a biological redox reaction in which an inorganic nitrogen compound is reduced. There are two mechanisms of reduction: (1) Assimilative denitrification and (2) Dissimilative denitrification.

(1) Assimilative denitrification refers to the reduction of nitrite or nitrate to ammonia for subsequent utilization in cell synthesis. Assimilative denitrification only occurs if no ammonia for synthesis is available, a situation unlikely to develop in municipal wastewater treatment systems, so that this process is not of any practical significance.

(2) Dissimilative denitrification refers to the redox reaction in which an inorganic nitrogen compound is reduced to a compound in which the nitrogen atom has a lower oxidation number*. The following sequence of reductions is possible:

$$NO_3^- \rightarrow NO_2 \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \qquad (2.23)$$

In each of these five steps one electron is transferred to the nitrogen atom, i.e. nitrogen serves as an electron acceptor and the oxidation number of nitrogen is reduced from a maximum of +5, in the nitrate ion,

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* The oxidation number of an atom is the charge (expressed in terms of the elementary charge, i.e. that of an electron) that can formally be attributed to a particular atom. For example in the nitrate ion, NO_3^- there are 3 x 2 = 6 negative charges due to the three oxygen atoms and as the ion is a monovalent anion the formal charge of the nitrogen ion is 6 - 1 = 5. Hence the oxidation number of nitrogen in NO_3^- is +5. In nitrite, NO_2^- there are only four negative charges due to oxygen and because this ion is also a monovalent anion the oxidation number of nitrogen in nitrite is +3. Hence for reduction of a nitrate ion to a nitrite ion 2 electrons are required to reduce the oxidation number of nitrogen from +5 to +3.

to zero in molecular nitrogen. Although it has been shown that the intermediate products are formed (Delwiche, 1956) it is generally accepted that reduction to nitrogen gas is predominant. Normally oxidised inorganic nitrogen in wastewater treatment systems is present almost exclusively as nitrate. Hence for all practical purposes denitrification can be considered as a biological redox reaction in which nitrate is reduced to nitrogen gas. The electron donor (or reductor) is biodegradable organic material and the redox reaction takes place in an anoxic environment and is mediated by facultative heterotrophic bacteria. An anoxic environment is characterised by the presence of nitrate (or nitrite) and the absence of dissolved oxygen in an aqueous solution. In an aerobic environment (i.e. an environment in which dissolved oxygen is present), oxygen is used as the final electron acceptor in the oxidation of biodegradable organic material. In an anoxic environment nitrate readily replaces oxygen as the final electron acceptor because the electron pathway for the transfer of electrons from the organic compound to the final electron acceptor is similar - the only difference is the final electron transfer from the cytochromes to oxygen or nitrate (Christensen and Harremoes, 1972). When the transfer is to nitrate instead of oxygen the specific enzyme oxygen reductase is replaced by the enzyme nitrate reductase to effect the final electron transfer. Studies of pure cultures of denitrifying organisms indicate that the presence of dissolved oxygen prevents the formation of the enzyme necessary for the final electron transfer to nitrate (Chang and Morris, 1972).

The half reactions of reduction of oxygen and nitrate can be written as:

$$\frac{1}{4}\,O_2 + H^+ + e \rightarrow \frac{1}{2}\,H_2O \qquad (2.24)$$

$$\frac{1}{5}\,NO_3^- + \frac{6}{5}\,H^+ + e \rightarrow \frac{1}{5}\,N_2 + \frac{3}{5}\,H_2O \qquad (2.25)$$

From Eqs (2.24 and 2.25) it may be noted that stoichiometrically $\frac{1}{4}$ of a mole of oxygen (i.e., 8g O_2) is equivalent to $\frac{1}{5}$ of a mole of nitrate (i.e. 2,7gN). Hence,

$$1mg\ NO_3^- - N \equiv 2,86mg\ O_2 \qquad (2.26)$$

In the nitrification process, stoichiometrically the oxygen requirement for oxidation of ammonia to nitrate is $4,57mg\ O_2 . mg\ N^{-1}$ (Table 2.3). In denitrification the reduction of 1mg of $NO_3 - N$ is equivalent to the reduction of $2,86mg\ O_2$. Hence, a fraction $2,85/4,57 = 0,625$ of the oxygen required for nitrification can be "recovered" as equivalent oxygen in the denitrification process.

This fraction can also be calculated readily by considering the changes in the oxidation numbers of the nitrogen atom to the processes of nitrification and denitrification. In the nitrification process, ammonium nitrogen (oxidation number -3) is converted to nitrate nitrogen (oxidation number +5); i.e. per nitrogen atom, 8 electrons are removed and transferred to the oxidant, oxygen. In the denitrification process, nitrate is converted to nitrogen gas (oxidation number 0) i.e. 5 electrons are transferred from the organic substrate to the nitrogen atom. Hence from the 8 electrons removed from the nitrogen atom in the nitrification process, 5 are recovered in the denitrification process, i.e. a fraction $\frac{5}{8} = 0,625$ of the electrons removed from the nitrogen atom in the nitrification process are "recovered" in the denitrification process. The electron transfers during nitrification and denitrification are shown schematically in Fig 2.4.

The half reactions of reduction of oxygen and nitrate (Eqs (2.24 and 2.25) respectively) may be coupled to the half reaction for oxidation of organic material. Assuming a general structure formula $C_xH_yO_z$ for organic material:

$$C_xH_yO_z + (2x - z)\ H_2O \longrightarrow x\ CO_2 + (4x + y - 2z)H^+ + (4x + y - 2z)e \qquad (2.27)$$

$$\frac{1}{4x+y-2z}\ C_xH_yO_z + \frac{2x-z}{4x+y-2z}\ H_2O \longrightarrow \frac{x}{4x+y-2z}\ CO_2 + H^+ + e \qquad (2.28)$$

Combining Eq (2.28) with (2.24) for the redox reaction, i.e. with oxygen as oxydant:

$$\frac{1}{4x+y-2z}\ C_xH_yO_z + \frac{2x-z}{4x+y-2z}\ H_2O + \frac{1}{4}\ O_2 \longrightarrow \frac{x}{4x+y-2z}\ CO_2 + \frac{1}{2}\ H_2O \qquad (2.29)$$

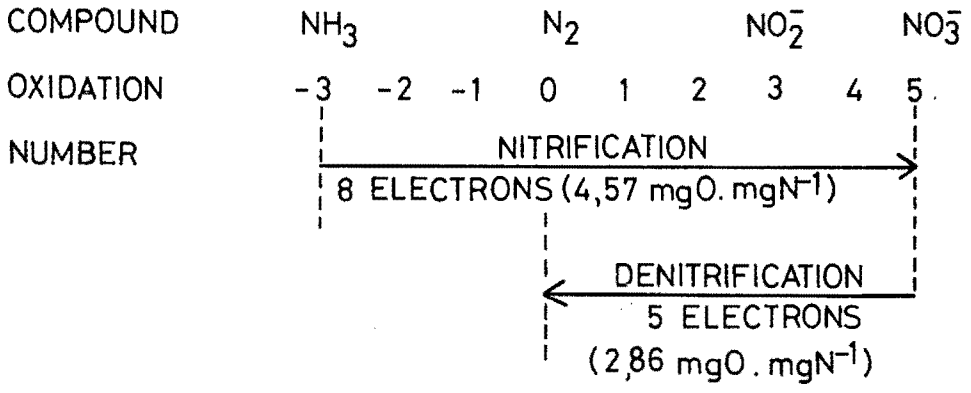

Fig 2.4 *Electron transfers in the nitrification and the denitrification reaction.*

Combining Eq (2.28) with Eq (2.25) for the redox reaction i.e., with nitrate as oxidant:

$$\frac{1}{4x+y-2z} C_xH_yO_z + \frac{2x-z}{4x+y-2z} H_2O + \frac{1}{5} NO_3^- + \frac{1}{5} H^+ \rightarrow \frac{x}{4x+y-2z} CO_2 + \frac{1}{10} N_2 + \frac{3}{5} H_2O \quad (2.30)$$

From a theoretical investigation, McCarty (1972) concluded that the free energy release associated with Eq (2.29) (i.e. oxidation of organic material with oxygen involving the transfer of 1 electron equivalent) is approximately the same as the free energy release associated with Eq. (2.30) (i.e. the oxidation of organic material with nitrate involving the transfer of 1 electron equivalent). Table 2.11 shows numerical values of the free energy release per electron equivalent for different organic materials when either oxygen or nitrate is used as oxidant. From Table 2.11 may be noted that Eq. (2.26) is not only valid for stoichiometric calculations but applies also approximately for calculations of the energy released. If it is assumed that the

energy requirement for cell synthesis in an anoxic environment is the same as that in an aerobic environment, it can be inferred that the cell growth yield coefficient from utilization of biodegradable organic material in an anoxic environment is essentially the same as that in aerobic environment.

From Eq (2.29) there is no consumption or production of hydrogen ions in the oxidation reaction of organic material with oxygen, i.e. there is no change in the alkalinity. (The formation of CO_2 does not affect the alkalinity; Loewenthal and Marais, 1976). From Eq (2.30) there is a consumption of 1 mole of hydrogen ions per mole of nitrate denitrified to nitrogen gas. Hence there is a stoichiometric alkalinity increase of 3,57 mg $CaCO_3$. mg $NO_3 - N^{-1}$. This is in good agreement with experimental values that have been reported (Scearce *et al*, 1978).

3.2 Requirements for denitrification

For denitrification to occur the following conditions are necessary:

1. Presence of a facultative bacterial mass.
2. Presence of nitrate in an aquatic solution.
3. Absence of dissolved oxygen.
4. Suitable environmental conditions for growth of micro-organisms.
5. Absence or limited concentration of toxic substances.
6. Presence of a suitable electron donor (energy source).

These factors will now be discussed in detail.

3.2.1 Facultative bacterial mass

The ability to denitrify is wide-spread among bacteria (Christensen and Harremoes, 1977). Dissimilative denitrification with end products N_2, NO and NO_2 has been established in numerous cases. The bulk of the bacterial mass in wastewater treatment is facultative, of which a significant fraction is capable of dissimilative denitrification.

Table 2.11 Comparison of energy transfer when oxygen or nitrate serves as final electron acceptor (after McCarty, 1972)

Energy Source	Energy transfer (kcal/electrol transfer)		
	Electron Acceptor		Ratio
	Oxygen	Nitrate	
1. Domestic wastewater	26,275	24,728	1,062
2. Methanol	27,640	26,093	1,059
3. Ethanol	26,267	24,720	1,062

Studies on denitrification tend to support the view that there is little difference between the bacterial masses in processes where nitrification only or nitrification-denitrification takes place. There is little merit therefore in attempting to analise the bacterial composition of the sludge in detail. A sludge generated under aerobic conditions, when subjected to the appropriate environmental conditions, will show a denitrifying capability immediately and will continue to do so subsequently without apparent change in reactivity.

3.2.2 Presence of nitrate

Normally the presence of nitrate implies nitrification as a pre-requisite for denitrification. The concentration of nitrate has little influence upon the denitrifying activity of suspended cultures; this has been reported by several authors (Balakrishnan and Eckenfelder, 1969; Moore and Schroeder, 1971; Sutton, Murphy and Yank, 1975). The relationship between denitrification activity and nitrate concentration can be expressed in terms of the Monod relationship with a very low half saturation value of about 0,1 mg NO_3 - N.ℓ^{-1} (Moore and Schroeder, 1971). For all practical purposes denitrification activity can be taken as independent of the nitrate concentration.

3.2.3 Absence of dissolved oxygen

It is generally accepted that the presence of dissolved oxygen in a mixed liquor inhibits denitrification. The mechanism of oxygen inhibition on the cellular level is known in some detail. Oxygen either represses the formation of the enzyme nitrate reductase or acts as an electron acceptor thereby preventing the reduction of nitrate (Christensen and Harremoes, 1977).

A major problem in quantifying the influence of the concentration of dissolved oxygen on denitrification is that concentration gradients develop in the flocs of microorganisms so that the DO concentration in the micro-environment around a cell may be quite different from the measured value in the bulk solution. Indeed, significant nitrate removal has been reported in aerobic reactors (Pasveer, 1965, and Matsché, 1971); this is likely to be due to the formation of micro-anoxic zones inside the flocs. In general dissolved oxygen concentrations of more than 0,2 (Carlson, 1970) to 0,5 $mg.\ell^{-1}$ (Christensen and Harremoes, 1977) significantly reduce denitrification activity in suspended sludge systems.

With regard to the definition of the state associated with the absence of oxygen, a usage of certain terms has arisen in nitrification-denitrification kinetics which differs from its biochemical counterpart: the term anoxic is defined to denote an environment in which there is an absence of oxygen but nitrate is present; the term anaerobic is defined to denote an environment in which both oxygen and nitrate are absent. (The biochemical usage of the term anaerobic denotes the absence of oxygen but the present of nitrates). Differentiation between anoxic and anaerobic states is necessary because the occurrence of an anaerobic state has been found to be a pre-requisite for inducing luxury biological uptake of phosphorus in nitrification-denitrification systems.

3.2.5 Environmental conditions

Among the most important environmental conditions for denitrification are the temperature and pH. Denitrification activity increases with increasing temperature up to an optimal of about 40°C. A temperature above about 50°C prevents denitrification from taking place

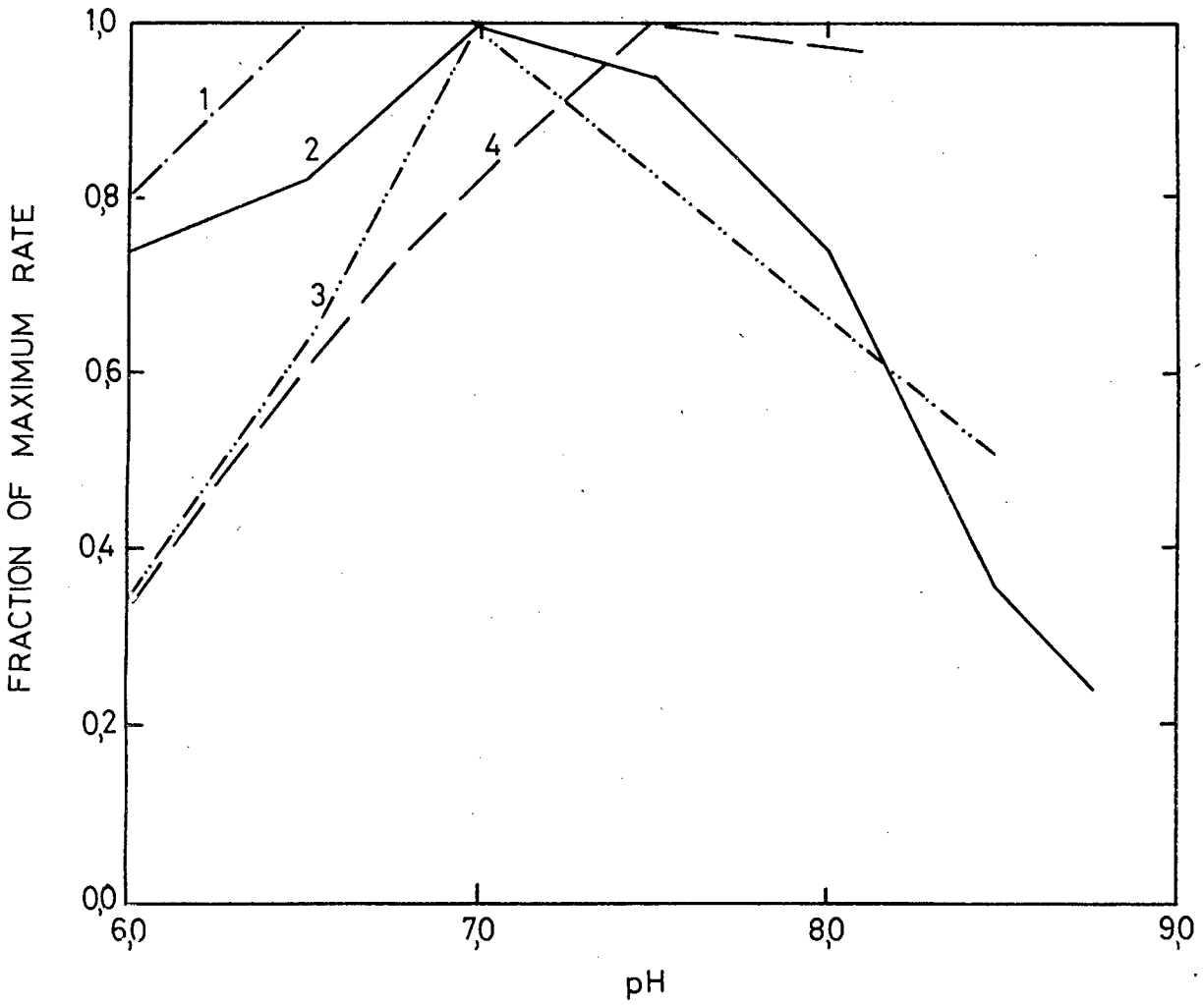

Fig. 2.5 *Influence of the pH on the rate of denitrification (after EPA, 1975).*

(1) : Mulbarger (1971).

(2) : Dawson and Murphy (1973).

(3) : Christensen and Harremöes (1972).

(4) : Clayfield (1974).

rise to inhibition are difficult to identify. Heavy metal ions usually have a relatively minor transitory effect but there appear to be organic substances that are extremely toxic. However usually nitrification is much more sensitive to inhibitory substances than the aerobic or anoxic organic material degradation and normally nitrification is ceased before inhibition of denitrification is detected. Generally if nitrification takes place, denitrification is also possible.

3.2.7 Presence of a suitable electron donor

Nitrification-denitrification systems can be classified according to the source of organic material that is utilized in the denitrification reaction. Organic material sources can be categorised as follows:

(a) Energy sources not present in the wastewater, i.e. an *external carbonaceous energy source* which is added at the denitrification stage of the process. Compounds used as an external energy source include methanol, methane, ethanol, acetone and acetic acid.

(b) Energy sources present in the influent wastewater, i.e. *an internal carbonaceous energy source* which enters the system with the wastewater.

(c) Energy sources which are *self-generated* within the system by the release of nutrient by the organisms as a result of death and lysis of a fraction of these.

The axis about which all biological denitrification investigations revolves is the energy source which serves as the electron donor in the denitrification process; the type of energy source defining to a large degree the process configuration for denitrification. The inter-relationship between energy source and process configuration, therefore, is of great importance.

3.3. Process configurations

3.3.1 External energy source

The denitrification process in which an external energy source is utilized forms part of the 'three sludge system' proposed by Barth, Bremmer and Lewis (1968) (see Fig 2.6).

This system consists of three completely separate stages. Each stage operates as a completely mixed activated sludge plant having its own reactor, settling tank, sludge mass and recycle. The first stage is aerobic and has a very short sludge age. The sole objective here is to remove the carbonaceous energy from the water. The effluent from the first stage passes to a second stage which operates at a long sludge age, and induces nitrification; the sludge mass generated consists principally of nitrifying organisms. The nitrified effluent passes to the third stage, the denitrification stage. Here, an organic source is fed to an activated sludge process, maintained free of oxygen, and organism metabolism takes place by the utilization of the NO_3^- as sole electron acceptor. Many different organic substances have been proposed as an energy source for denitrification; the most used being methanol (Christensen and Harremoes, 1977).

A modification to the 'three sludge system' is the 'two sludge system' proposed by Johnson and Vania (1971). In this system the first two stages of the 'three sludge system' are combined into one stage. The size of reactor for combined carbonaceous oxidation and nitrification is probably the same as that for the nitrification reactor in the 'three sludge system'. The 'two sludge system' requires only two settling tanks and hence tends to be the more economical system.

In the three and two sludge systems the denitrification reactor kinetically behaves in exactly the same fashion as an aerobic reactor. The theory of this process has been developed by Stensel, Loehr and Lawrence (1973). The forms of the equations for sludge generation and nitrate utilization are identical to those for the aerobic activated sludge system as developed by Lawrence and McCarty (1970). The only differences are in the values allocated to the various kinetic constants. However, there is a significant difference in the

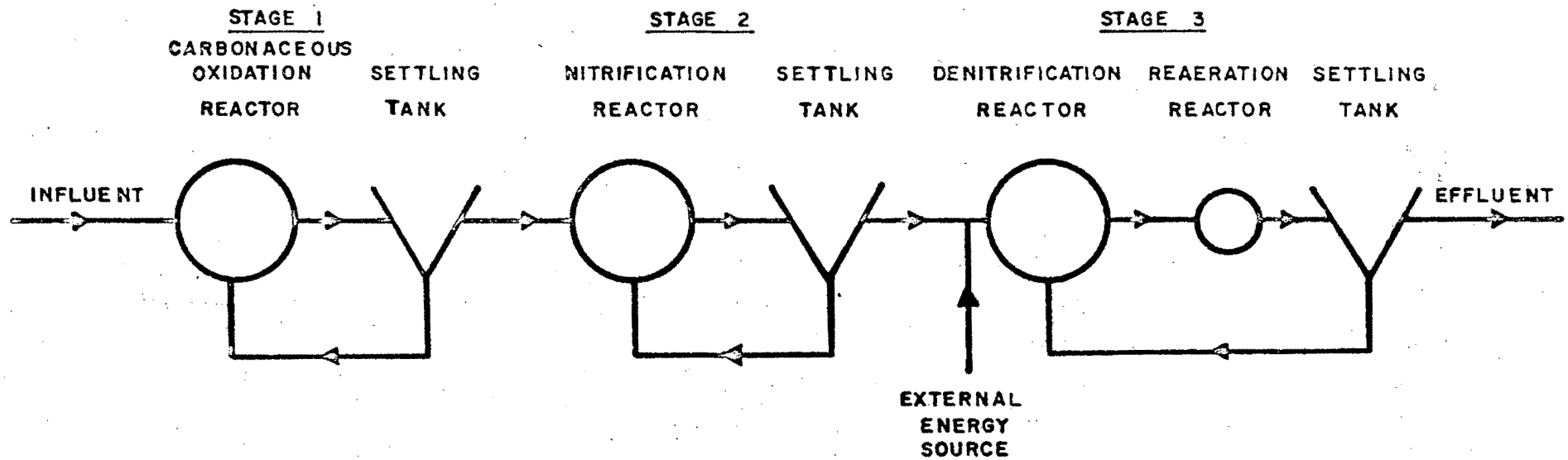

Fig. 2.6 *The three sludge systems for denitrification with an external energy source.*

objectives between an aerobic and a denitrification activated sludge process. In the aerobic process the organic energy supply is known and it is necessary to determine the oxygen supply necessary to metabolise the energy. In the denitrification process the nitrate (i.e. the equivalent oxygen supply) is known and it is necessary to determine the organic energy supply in order to utilise all the nitrate. Although theoretically it is possible to regulate the energy supply to be just sufficient to utilise the nitrate, practically perfect matching is difficult to obtain and nitrate or organic material will be present in the effluent. To reduce excess COD a flash aeration reactor is interposed between the denitrification reactor and the settler. The flash aeration reactor also assists in stripping nitrogen bubbles from the sludge flocs to promote liquid solid separation in the settler. Sludge flocs are light and may have bad settling characteristics.

Due to the specific natures of the energy source and the electron acceptor respectively, the organism mass that develops in the denitrification reactor is also highly specific. Two and three sludge systems have been put into operation at full scale level (Horstkotte, Niles, Parker and Caldwell, 1978; Mulbarger, 1972). The plants apparently are reliable and give good performance but operational costs are high, or even prohibitive, due to the high cost of the external energy source. Stoichiometrically the mass of methanol per mg of nitrate nitrogen reduced can be determined as follows:

1 mg CH_3OH = 1,5 mg COD

1 mg NO_3-H = 2,86 mg equiv. O_2

Hence

1 mg NO_3-H = 2,86/1,5 = 1,92 mg CH_3OH

If account is taken of the energy incorporated in the sludge mass, 1 mg NO_3 requires 2,2 - 2,5 mg of CH_3OH depending on the sludge age (Christensen and Harremoes, 1977).

3.3.2 Self generated energy source

The nitrification-denitrification systems in which endogenous energy release provides the energy source for denitrification was first proposed by Wuhrmann (1964). It is one of the group of single sludge nitrification-denitrification systems. A schematic presentation of the process is shown in Fig. 2.7. It consists of two reactors in series, the first aerobic and the second anoxic. The influent is discharged to the first reactor where aerobic growth of both the heterotrophic and nitrifying organisms takes place. Provided the sludge age is sufficiently long and the aerobic fraction of the system is adequately large, nitrification is complete in the first reactor.

The mixed liquor from the aerobic reactor passes to the anoxic reactor, also called the postdenitrification reactor, where it is kept completely mixed by stirring but with no aeration. The outflow from the anoxic reactor passes through a settling tank and the underflow is recycled back to the aerobic reactor.

Energy release by organisms due to endogenous mass loss provides the principal energy source for denitrification in the anoxic reactor. However, the rate of release of energy is low, so that the rate of denitrification is also low. Consequently in order to obtain a meaningful reduction of the nitrate concentration in the anoxic reactor the anoxic fraction of the system must be large and this may cause a breakdown of the nitrification process. Thus although theoretically the system has the potential to remove all the nitrate, from a practical point this usually is not possible as the anoxic volume fraction needs to be so large that the conditions for nitrification cannot be satisfied.

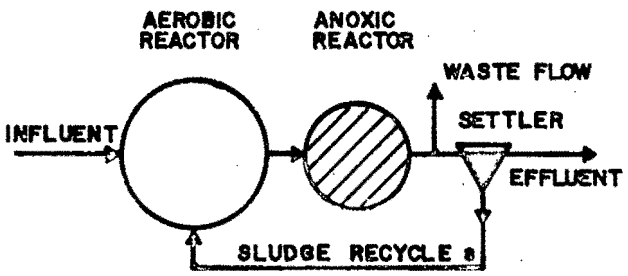

Fig.2.7 *The Wuhrmann Configuration.*

3.3.3 Internal energy source

Processes based on an internal energy source also belong to the single sludge group. In these processes the biodegradable material in the influent is used as the electron donor for reduction of nitrate. There are two different processes based on this principle:

(1) A multireactor process in which the reactor receiving the influent is unaerated, i.e. continuously anoxic.

(2) A single reactor system with alternating periods during which the mixed liquor is aerated followed by periods during which the mixed liquor is not aerated.

(1) Multireactor process.

Ludzack and Ettinger (1962) were the first to propose a process utilising the biodegradable material in the influent as an

energy source for denitrification. A schematic presentation of this system is shown in Fig 2.8. It consists of two reactors in series, *partially* separated from each other. The influent is discharged to the first reactor which is maintained in an anoxic state by stirring without aeration. The second reactor is aerated and nitrification takes place. As there is only partial separation, the mixed liquor in the first reactor is in communication with that in the second reactor. Due to the mixing action in both reactors an interchange of the nitrified and anoxic liquors is induced, and the nitrate entering the anoxic reactor is reduced to nitrogen gas. Ludzack and Ettinger reported that the process gave variable denitrification results, probably due to the lack of control of the interchange of the contents of the two reactors.

Barnard (1972) proposed an improvement of the Ludzack-Ettinger process, by completely separating the anoxic and aerobic reactors, recycling the underflow from the settler to the anoxic instead of the aerobic reactor, and providing a recycle from the aerobic to the anoxic reactor. The resulting modified Ludzack Ettinger system is shown in Fig 2.9. The modifications allow a significantly improved control over the process performance. The high energy source discharged to the anoxic reactor (also called the predentrification reactor) gives rise to a high rate of denitrification and a substantially higher reduction of nitrate than in the Wuhrmann process, even when the predenitrification reactor is substantially smaller than the post-denitrification reactor. High removal efficiencies have been reported (Stern and Marais, 1973; Nicholls, 1975; Sutton *et al*, 1979). A problem with the predenitrification system is that complete denitrification cannot be achieved because part of the nitrates from the aerobic reactor are not recycled, but discharged directly with the effluent. The nitrate removal will be affected by the recycle rate to the anoxic reactor. Low nitrate removal efficiencies will be obtained if low recycle ratios are employed.

(2) Single reactor alternating process.

The alternating process in its simplest form is a single

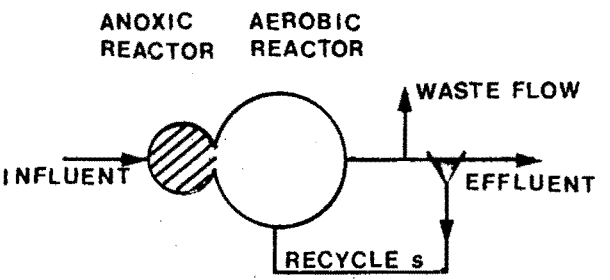

Fig. 2.8 *The Ludzack Ettinger denitrification system.*

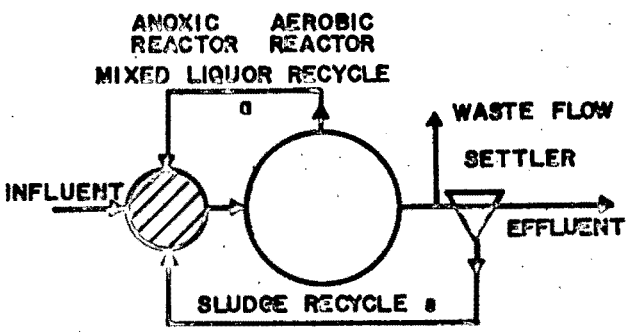

Fig. 2.9 *The Modified Ludzack Ettinger (MLE) or pre-denitrification configuration.*

reactor system receiving a continuous influent flow. Periods of aeration (during which nitrification of the influent TKN takes place) are followed by periods with no aeration (during which denitrification of the nitrified TKN takes place). It is obvious that in such a system neither nitrification nor denitrification can be complete: During the aerobic period there is an increasing concentration of nitrate in the reactor (and hence in the effluent) and during the anoxic period there is an increasing concentration of ammonia derived from the influent and endogenous mass loss.

The performance of the system may be improved by the operational method suggested by Christensen (1974) and Klapwijk (1974) and schematically presented in Fig 2.10. The single reactor system is substituted by a two reactor system and operated in cycles of four phases: In the first phase the reactor receiving the influent (the first reactor) is not aerated and the second reactor is aerated; in the second phase neither one of the reactors is aerated; in the third phase the influent flow is introduced in the second reactor, whereas the effluent is abstracted from the first reactor. Again the reactor receiving the influent is not aerated, whereas the other is aerated. The reason for this rather complex operational mode is that it was thought that not enough nitrates would be transported to the anoxic reactors such as in the modified Ludzack Ettinger configuration (Christensen and Harremoes, 1977). However this conclusion does not appear to be valid; with proper design and operation very low effluent nitrate concentrations have been reported by several authors operating modified Ludzack Ettinger or Bardenpho configurations (see next section).

3.3.4 Combination of self generated and internal energy source - The Bardenpho configuration

In order to overcome the deficiency of incomplete denitrification in the modified Ludzack-Ettinger process, Barnard (1972) proposed combining this process with that of Wuhrmann. A schematic presentation of the combined or *Bardenpho* system is shown in Fig 2.11. Barnard considered that the low concentration of nitrate discharged

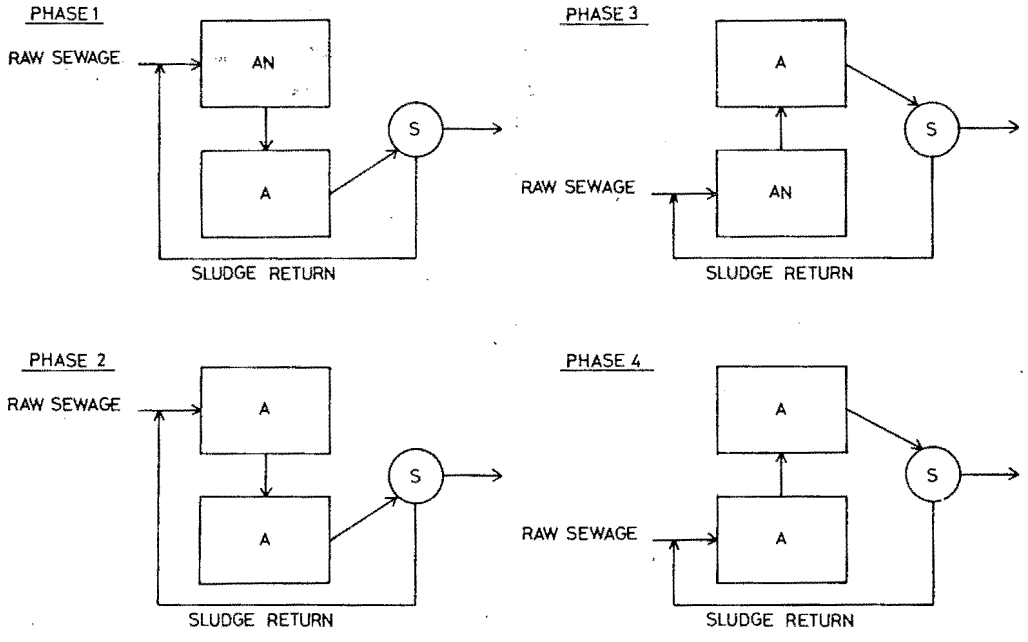

Fig.2.10 *Alternating process operated in four phases(after Christensen,1975).*

from the aerobic to the postdenitrification reactor will be denitrified to produce a relatively nitrate free effluent. In order to strip the nitrogen bubbles generated in the postdenitrification reactor and attached to the sludge flocs Barnard proposed placing a flash aeration reactor before the final settling tank. The flash aeration reactor was also considered necessary to nitrify the ammonia released during the sludge residence time in the postdenitrification reactor. In order to reduce the possibility of flotation of sludge in the settler due to denitrification of residual nitrate the sludge accumulation in the settler was to be kept to a minimum. This was achieved by having a high recycle rate from the settler, approximately equal to the mean influent flow.

Although in concept the Bardenpho process has the potential for complete removal of nitrate, in practice this is not always possible as the anoxic reactors cannot be increased *ad lib* for the reason that nitrification ceases in processes with an inordinately large anoxic mass fraction.

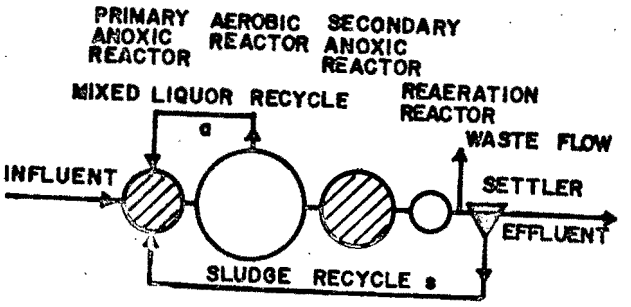

Fig.2.11 *The Bardenpho configuration.*

The design procedure of the Bardenpho system was essentially empirical and did not allow rational optimization of recycle ratios, the relative reactor sizes of the anoxic zones or precise estimates of the nitrogen removal. There was also considerable uncertainty as to the influence of the anoxic zones on nitrification. Complete or near complete denitrification has been obtained by different authors (Barnard, 1973; Heide, 1977; Heideman, 1979).

3.4 Multi sludge versus single sludge denitrification systems

When multisludge systems (i.e. two and three sludge systems) are compared with single sludge systems the following advantages of single sludge systems can be listed.

(1) lower operational cost (energy source is free)
(2) oxygen for nitrification can partially be recovered
(3) alkalinity increase due to denitrification is effectively utilised.

(1) If complete denitrification in a single sludge system is possible, there is no need for an external energy source. The daily production of TKN and COD are respectively about 10 g TKN and 100 g COD per inhabitant. About 20 per cent of the TKN is required for synthesis and the remaining 80 per cent is nitrified, so that the production of nitrate is of the order of 8 g NO_3 - N. habitant^{-1}.day^{-1}. In the denitrification reactor of a multisludge system the methanol requirement is approximately 2,5 g CH_3OH.g NO_3 - N^{-1} denitrified. Consequently the consumption of methanol will be about 20 g CH_3OH. d^{-1}.habitant^{-1} or 50ℓ CH_3OH.year^{-1}. habitant^{-1}. In single sludge systems many investigators have observed a high removal efficiency of nitrate and in instances where this high efficiency could not be obtained,the causes often were due to inadequate design features such as (a) insufficient anoxic sludge mass fraction, (b) non optimal location of the anoxic reactor in a series configuration, (c) insufficient recycle of nitrate to the anoxic zone, (d) too short a sludge age, or, a combination of these. From the literature,however,it can be established with certainty that with appropriate design and operation a high percentage of nitrogen can be removed from municipal wastewaters by nitrification-denitrification single sludge systems, using the influent biodegradable organic material as the energy source and possibly the self generated energy from the sludge.

If complete denitrification in a particular single sludge system is desired and cannot be obtained using only these energy sources,it is possible to add an external energy source in an auxiliary capacity in such a proportion that complete removal of nitrate is obtained. If an additional energy source is necessary the required amount will only be a small fraction compared to that required to obtain a comparable nitrate reduction in a multisludge system, thus reducing significantly the operational costs of the treatment plant.

(2) The oxygen requirement for nitrification is about 25 to 35 percent of the total oxygen requirement depending on the influent TKN/COD ratio and the sludge age. If complete denitrification in a single sludge system is achieved,a fraction 0,625 of the oxygen requirement for nitrification can be recovered, representing about 20 per cent

of the total oxygen requirement. In multisludge systems the potential advantage of recovery of equivalent oxygen is not utilized.

(3) When low alkalinity wastewaters are treated in a multisludge system it is often necessary to add alkalinity (lime) to the nitrification reactor to counter the alkalinity lost in the nitrification process of 7,14 mg $CaCO_3$.(mg NH_4 - N)$^{-1}$. In the denitrification reactor up to half of the alkalinity lost in the nitrification process can be recovered (Eqs (2.1 and 2.30)) but in multisludge systems, because there is no feedback from the denitrification to the nitrification reactor, this recovery cannot be used beneficially.

There is, however, one advantage that the multisludge systems may exhibit above single sludge systems: In single sludge systems nitrification does not take place when the sludge is in an anoxic environment. This implies that in an anoxic/aerobic system the sludge age required for efficient nitrification must be longer than that required in the nitrification reactor of a multisludge system, and if a large anoxic sludge fraction in the single sludge system is necessary (which usually will be the case), this difference is significant. As the reactor volume increases with increasing sludge age, it may be possible that the total reactor volume of the three or two sludge systems is smaller than that of a single sludge system. However, this possible advantage of a multisludge system is very unlikely to compensate for the serious disadvantages that are inherent in these systems as listed above. The discussion of nitrification and denitrification systems will therefore be limited to the various configurations of the single sludge system.

3.5 Denitrification kinetics in single sludge systems

3.5.1 Experimentally based denitrification model

Carlson (1971) and Christensen and Harremoes (1972) suggested that the kinetic reaction describing denitrification by the activated sludge mixed liquor can be expressed by :

$$dN/dt = - KX_V \qquad (2.31)$$

and having a plug flow regime in the respective anoxic reactors. Experiments were carried out, using only settled or raw municipal sewage, over a range of sludge ages, sludge concentrations, influent COD concentrations and recycle ratios at temperatures between 14°C and 20°C. A schematical representation of the experimental set-up is show in Figs. 2.12 and 2.13 for the modified Ludzack Ettinger and the Wuhrmann configurations respectively.

From an analysis of all the experimental data reported by Marais and his co-workers, it would appear that the concentration of volatile solids in the anoxic/aerobic systems did not differ distinguishably from that in comparable completely aerobic systems and, that it may be calculated theoretically using the formulations for the different components in the volatile solids, developed by Marais and Ekama (1976). In their model the mixed liquor volatile suspended solids (MLVSS) mass consists of three fractions:

(1) Active sludge, X_a, i.e. the live organism concentration that can utilize biodegradable organic material.

(2) Endogenous residue, X_e. When active cells die a fraction of their mass is unbiodegradable particulate material and accumulates in the system as endogenous residue; the biodegradable fraction is used by active cells.

(3) Inert material, X_i. This fraction is derived from unbiodegradable particulate material in the influent and accumulates in the sludge mass by enmeshment (bioflocculation).

The following equations were derived by Marais and Ekama (1976) for a single reactor aerobic system:

$$X_a = (1 - f_{us} - P.f_{up}).S_{ti}.Y_h.R_s/((1 + b_{hT}R_s)R_h) \quad (2.32)$$

$$X_e = f.b_{hT}.R_s.X_a \quad (2.33)$$

$$X_i = f_{up}.R_s.S_{ti}/R_h \quad (2.34)$$

$$X_v = X_a + X_e + X_i \quad (2.35)$$

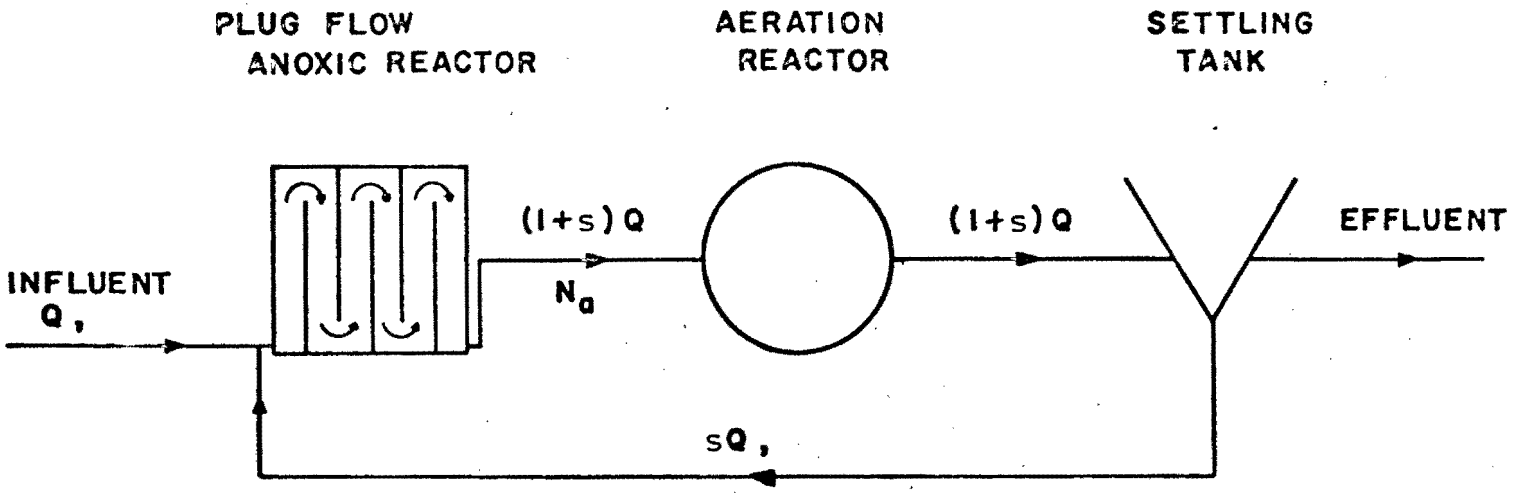

Fig.2.12 *Process configuration for pre-denitrification using an anoxic plug flow reactor.*

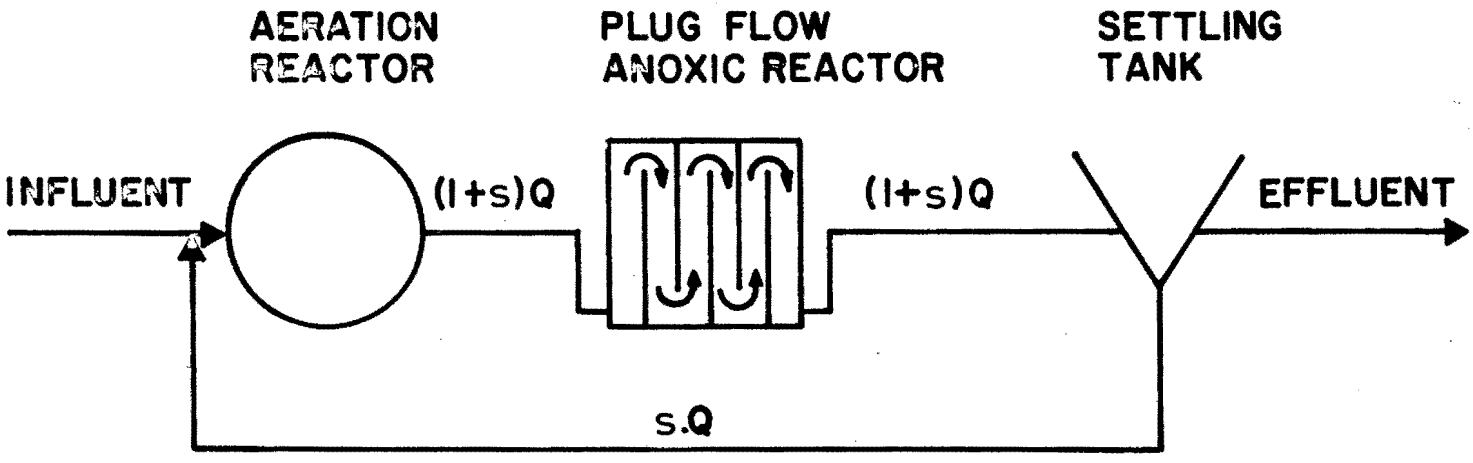

Fig.2.13 *Process configuration for post-denitrification using an anoxic plug flow reactor.*

where

X_a = active sludge concentration (mg VSS.ℓ^{-1})
X_e = endogenous residue (mg VSS.ℓ^{-1})
X_i = inert sludge fraction (mg VSS.ℓ^{-1})
X_v = total volatile solids concentration (mg VSS.ℓ^{-1})
f_{us} = unbiodegradable soluble influent COD fraction (mg COD,mg COD^{-1})
f_{up} = unbiodegradable particular influent COD fraction (mg VSS.mg COD^{-1})
P = COD/VSS ratio = 1,48 mg COD.mg VSS^{-1}
S_{ti} = Total influent COD (mg COD.ℓ^{-1})
Y_h = yield coefficient for heterotrophic sludge growth.
= 0,45 mg VSS.mg COD^{-1}
b_{ht} = endogenous respiration rate constant
= $0,24\ (1,029)^{T-20}$
R_s = sludge age or solids retention time
R_h = hydraulic retention time
f = unbiodegradable fraction of dead bacteria.
= 0,2 mg VSS.mg VSS^{-1}.

For convenience the expression for the active sludge concentration is rewritten as:

$$X_a = C_r \cdot S_{bi}/R_h \qquad (2.36)$$

where

$$C_r = Y_h R_s / (1 + b_{hT} R_s) \qquad (2.37)$$

S_{bi} = biodegradable influent COD concentration
= $(1 - f_{us} - P.f_{up}).S_{ti}$

The model by Marais and Ekama will be discussed in detail in Chapter 3.

From Eqs (2.32 to 2.35) the concentration of volatile solids depends upon the composition of the influent COD, i.e. on the fractions of unbiodegradable soluble (f_{us}) and particulate (f_{up}) material in the influent. For example Marais and Ekama found that in unsettled municipal waste flows in South Africa, f_{up} = 0,09 mg VSS.mg COD^{-1}. This value however may vary depending on the customs prevalent in the

country. This can be demonstrated by comparing the response of activated sludge plants in North America with those in South Africa. In South Africa domestic waste flows are derived from washing and sanitation usage; disposal of solid organic wastes to the sewer is prohibited by regulations. In North America the practice of disposal of garbage by grinding and discharge to the sewers is common and is one of the factors contributing to the very high per capita production of COD measured there. (In the U.S.A. 150 to 190 g COD.person^{-1}.day^{-1} against only 100 to 110 g COD.person^{-1}.day^{-1} in South Africa). Most of the garbage material is of a particulate nature and a considerable fraction is unbiodegradable. Consequently one can expect the unbiodegradable particulate fraction of the organic material in such sewages, f_{up}, to be high.

The value of f_{up} can be determined by analysing the response of an activated sludge plant to a particular sewage flow. This is illustrated by the following analyses of the results from two investigations in Canada and in the U.S.A.

Sutton *et al* (1979) carried out nitrification-denitrification experiments in single sludge systems using both pre- and post-denitrification configurations in completely mixed series reactor systems at pilot plant scale. The influent was raw sewage from Burlington, Ontario, Canada. The experiments were carried out at three different temperature ranges: 7 to 8°C, 14 to 16°C and 24 to 26°C. The sludge age was varied between a minimum value of 3 and a maximum value of 35 days and was controlled by sludge wastage from the underflow recycle.

In order to analyse the data the following procedure was used: The reported experimental ratios (mass of volatile sludge in the system): (mass of COD daily applied), i.e. (MX_v / MS_{ti}), were plotted as a function of sludge age for the different temperature ranges and are shown in Figs. 2.14 a, b and c. All the reported data using only raw sewage as influent are included in these figures. The theoretical ratio of (mass of sludge): (mass of daily applied COD), m_{xv}, can be expressed as follows:

$$m_{xv} = MX_v/MS_{ti} = (MX_a + MX_e + MX_i)/MS_{ti}$$

$$= Y_h R_s(1 + fb_{hT} R_s) (1 - f_{us} - P.f_{up})/(1 + b_{hT} R_s) + f_{up}.R_s \qquad (2.38)$$

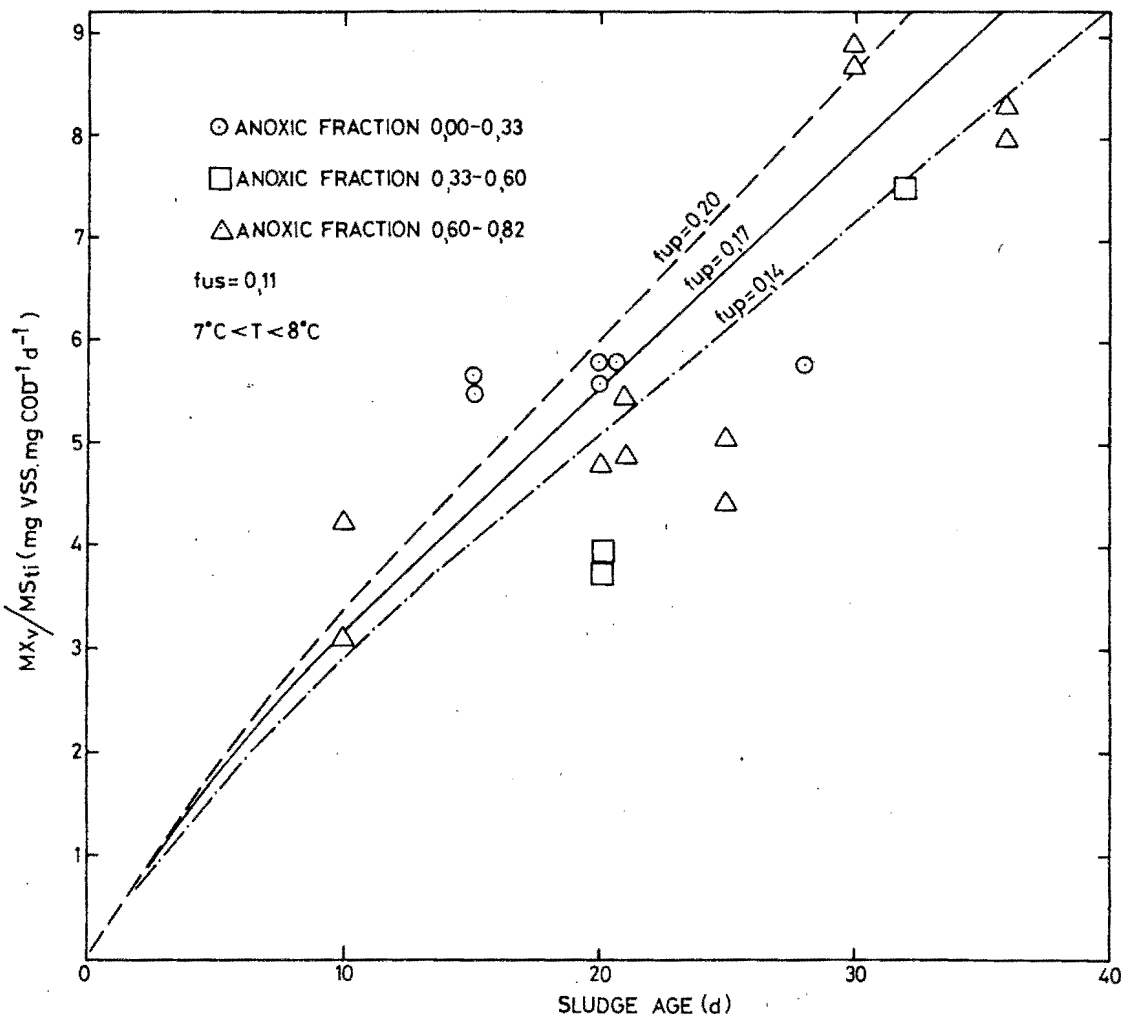

Fig 2.14a *Experimental and theoretical MX_v/S_{ti} ratios versus sludge age at T=(7,5±0,5)°C (experimental data by Sutton et al, 1979).*

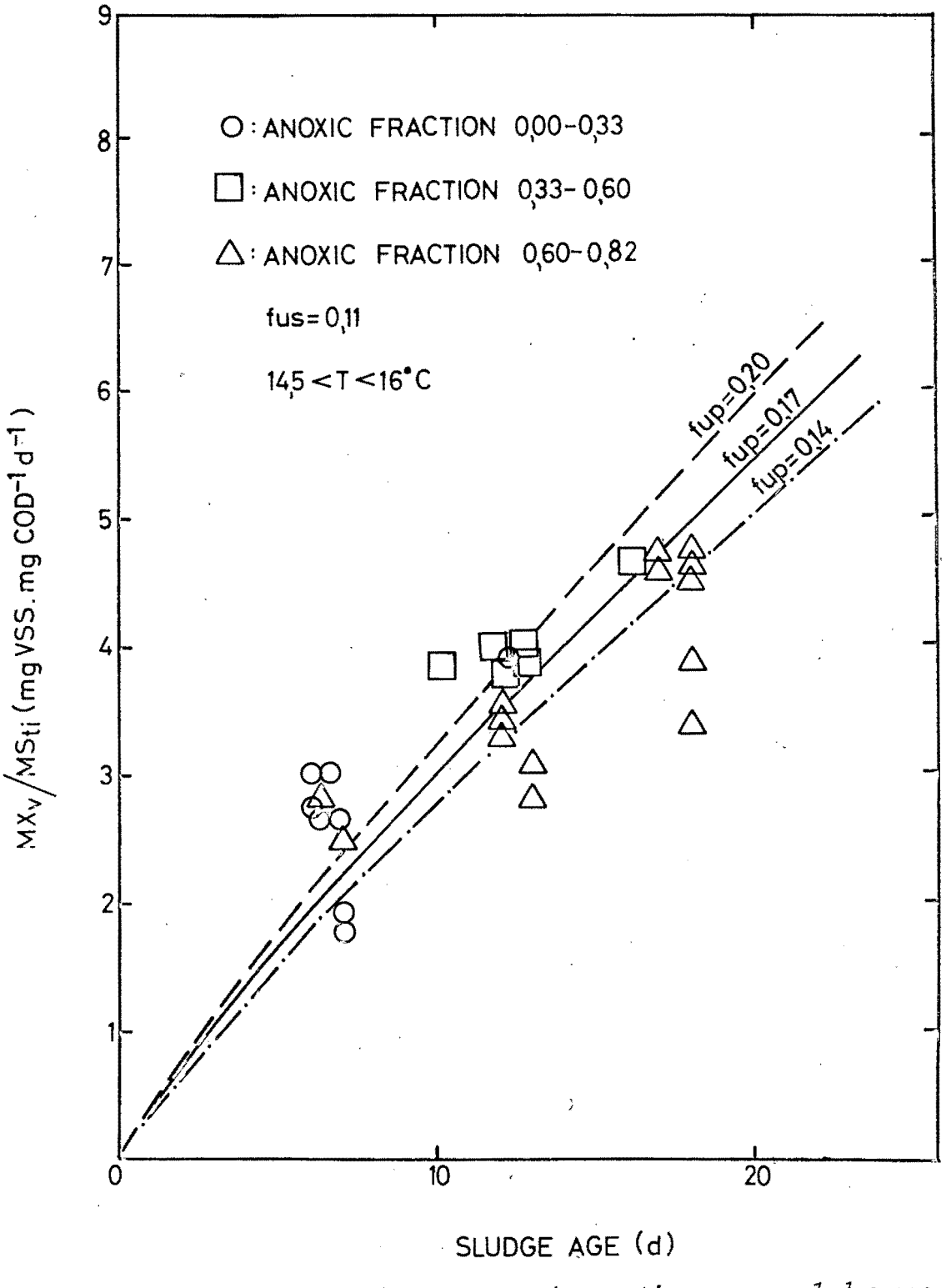

Fig.2.14b *Experimental and theoretical MX_v/S_{ti} ratios versus sludge age at T=(15±1) °C (Experimental data by Sutton et al, 1979).*

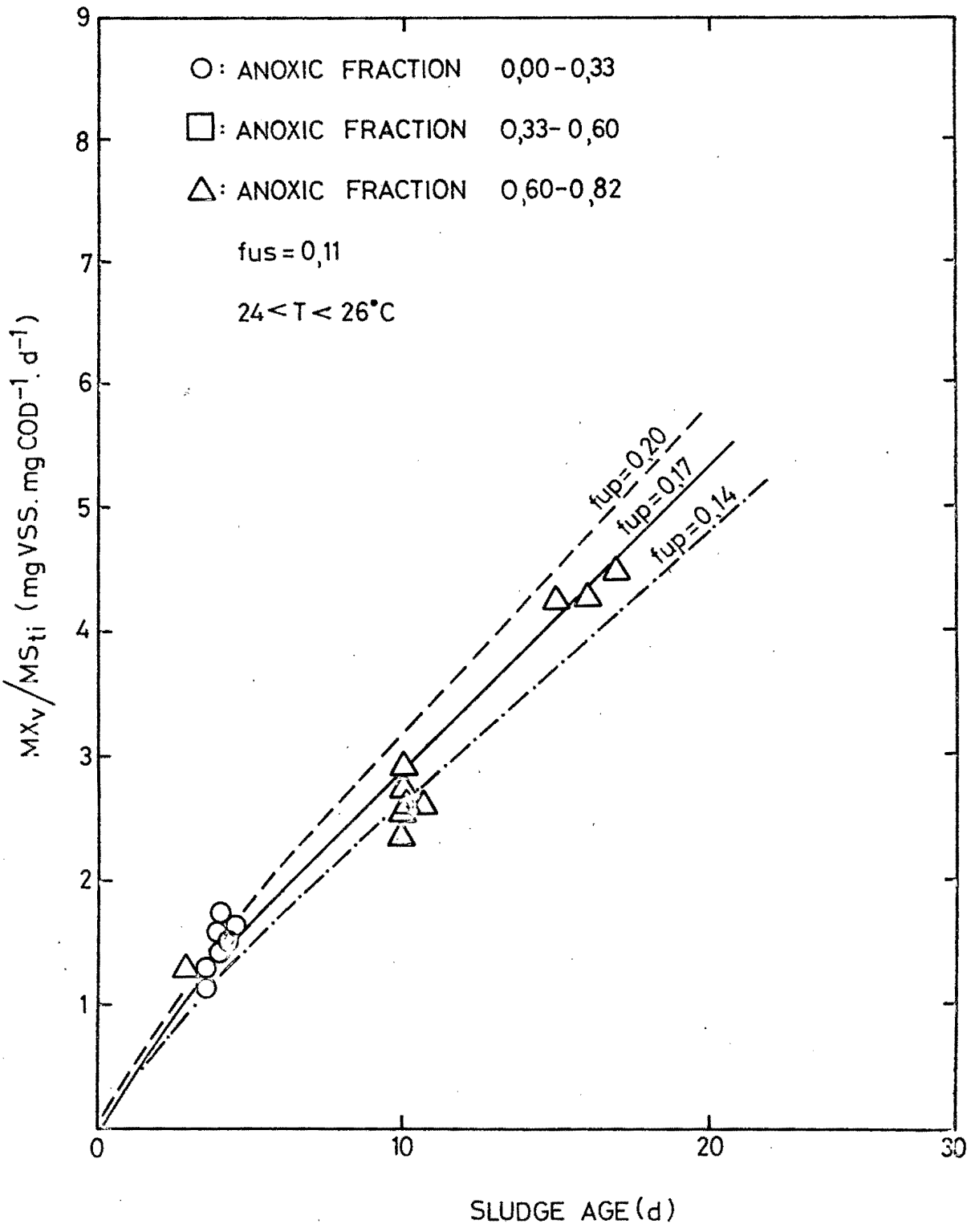

Fig.2.14c *Experimental and theoretical MX_v/S_{ti} ratios versus sludge age at T=(25±1)°C(experimental data by Sutton et al, 1979).*

where

MX_v = mass of volatile solids in the system = $V.X_v$

MX_a = mass of active sludge in the system = $V.X_a$

MX_e = mass of endogenous residue in the system = $V.X_e$

MX_i = mass of inert material in the system = $V.X_i$

MS_{ti} = mass of COD fed daily to the system

= $Q.S_{ti}$

V = total reactor volume

In Eq (2.38) the value of f_{us} can readily be estimated from the average effluent COD and was determined at f_{us} = 0,11 mg COD.mg COD^{-1}, which is a value not greatly different from that observed for domestic wastes in South Africa.

An estimate of f_{up} can now be made as follows: Calculate the theoretical $mx_v = MX_v/MS_{ti}$ ratio for different values of f_{up} as a function of the sludge age for the different temperatures (7,5°C, 15°C and 25°C) using Eq (2.38) and plot these as curves in Fig 2.14 a, b and c. The best overall fit between experimental and theoretical mx_v values is obtained for f_{up} = 0,17 mg VSS.mg COD^{-1}. The spread in the experimental data is possibly due, at least partly, to the fact that the sludge was wasted from the underflow which makes it very difficult to accurately maintain a specified sludge age. Also the f_{up} value usually varies randomly around the mean value.

The inert particulate fraction of f_{up} = 0,17 mg VSS.mg COD^{-1} (equivalent to $P.f_{up}$ = 1,48.0,17 = 0,25 mg COD.mg COD^{-1}) is very high when compared with the values obtained by Ekama and Marais (1976) in South Africa for unsettled municipal wastes (0,09 mg VSS.mg COD^{-1} or 0,13 mg COD.mg COD^{-1} respectively). The most likely reason for this difference is the use of garbage grinders in North America.

Heideman (1979) using *settled* sewage from Washington D.C. in a pilot plant postdenitrification system reported data on sludge production in this system. The system had an anoxic sludge mass fraction of about 66 per cent. In Fig 2.15 a similar plot to Fig 2.14 is made for the data reported by Heideman (1979). From the average effluent COD the value of f_{us} was established at f_{us} = 0,11 mg COD.mg COD^{-1}. The best correlation between experimental data and theoretical predictions

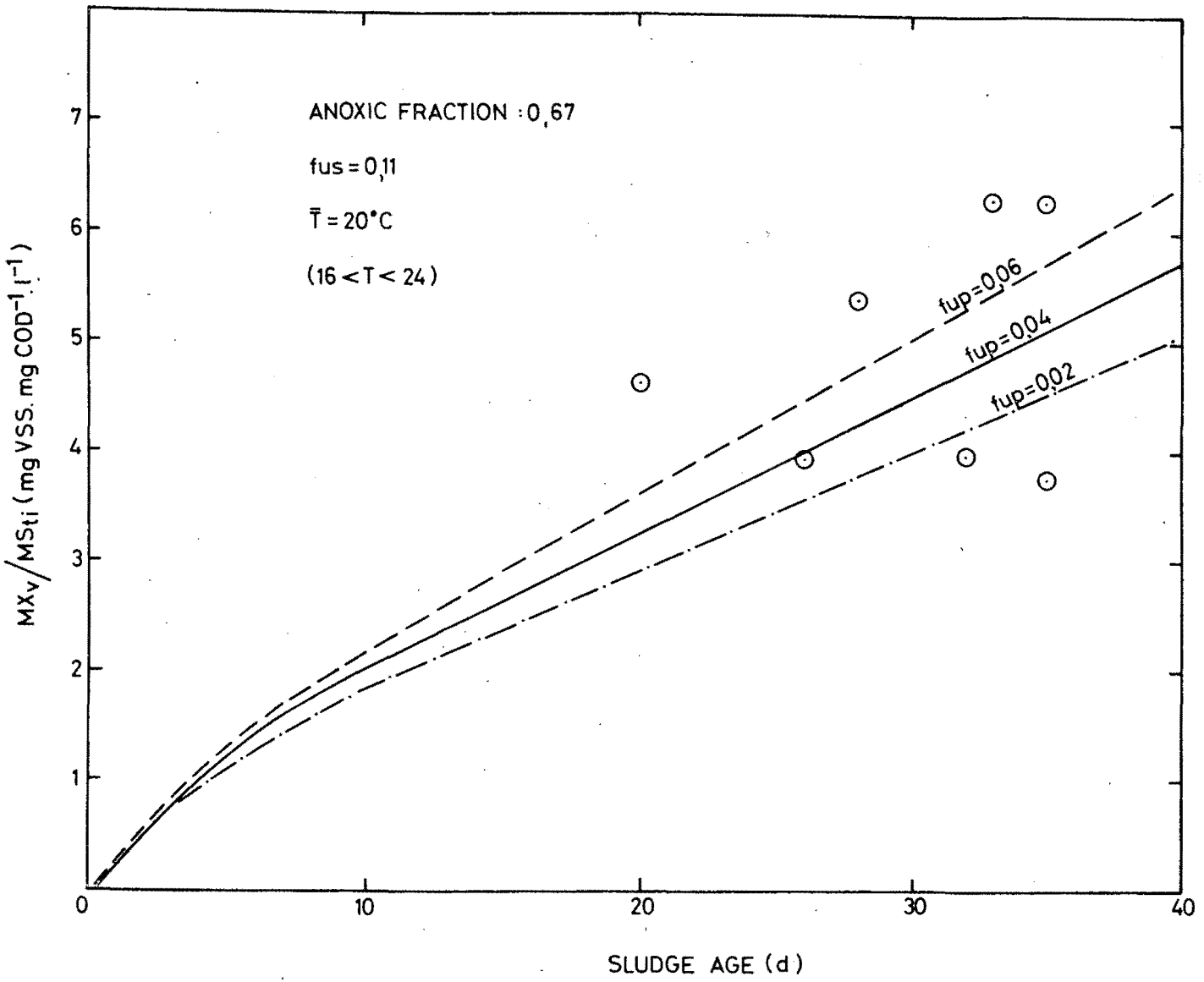

Fig.2.15 *Experimental and theoretical MX_v/S_{ti} ratios versus sludge age at T=(20±4) °C (experimental data by Heideman, 1979).*

was obtained for f_{up} = 0,04 mg VSS.mg COD^{-1}. This value is in accordance with the range of f_{up} values reported by Marais and Ekama (1976) for settled municipal sewage (0,02 < f_{up} < 0,05). The spread of the experimental data in Fig 2.15 is probably due to the fact that sludge wastage was discontinuous.

The analyses above presupposes the applicability of the theory developed by Marais and Ekama (1976). There is little that can be done to check this positively without an extensive investigation using the respective waste flows. However powerful support for this approach is obtained by investigating the denitrification behaviour; in Section 3.5.4 it will be shown that if the formulation by Marais and Ekama is accepted, the denitrification behaviour observed during the two investigations discussed above is consistent with the theory.

The data reported by Sutton *et al* (1979) also allows a check to be made whether the anoxic sludge mass fraction significantly influences the sludge production. The anoxic sludge mass fractions in the single sludge units operated by Sutton et al (1979) were varied from 28 to 82 per cent of the total sludge mass. In plotting the reported experimental data in Figs 2.14 a, b and c, these were divided into those obtained from systems with anoxic sludge mass fraction of 33 per cent or less, those between 33 and 60 per cent and those with 60 per cent or more. From Figs 2.14 a, b and c it would appear that the size of the anoxic sludge mass fraction did not measurably affect the sludge mass generated in these systems. The same conclusion was made by Stern and Marais (1974); Marsden and Marais (1976) and Wilson and Marais (1976). It would therefore appear that the sludge production in single sludge anoxic/aerobic systems does not deviate distinguishably from that in comparable completely aerobic systems and that the formulations by Marais and Ekama for total sludge concentration and the concentrations of active, endogenous and inert fractions are also applicable to anoxic/aerobic systems. It is very likely that this equivalence does not hold for very high anoxic sludge mass fractions but tests by Arkley and Marais (1981) indicated that no significant difference is observed for anoxic sludge fractions of up to 70 per cent at a sludge age of 20 days and a temperature of 20^{o}C.

A major difficulty in the interpretation of experimental data reported in the literature is that it is usually obtained at one sludge age (which is often not given) and one temperature. If data for only one sludge age is reported, it is not possible to establish whether the formulation of Marais and Ekama is applicable because from the data, at one particular sludge age, it is always possible to choose a value for f_{up} such that the observed sludge production is equal to the theoretical value predicted from Eq (2.38). Other indications of active sludge response, such as determinations of the carbonaceous oxygen uptake rate in the aerobic reactor, are needed. Ideally, data should be obtained from processes operating over a range of sludge ages as wide as possible, say from 2 to 30 days. In all cases where sufficient data is available, (i.e. in the work reported by Marais *et al*, Sutton *et al* and Heideman) it is possible to establish that the sludge production can be expressed in terms of the formulation developed by Marais and Ekama (1976). It will be accepted that this formulation is generally applicable for single sludge nitrification-denitrification systems.

3.5.3 Substrate utilization and associated nitrate removal

In the previous section it was shown that at any sludge age the sludge production in anoxic/aerobic systems is not distinguishably different from that in purely aerobic systems. It is inferred therefore that the yield coefficient, Y_h, and the endogenous respiration constant, b_{hT}, retain the same values in an anoxic environment as in an aerobic environment. In Chapter 3 it will be shown that in an aerobic environment the metabolic processes of substrate utilization by active organisms result in the oxidation of a fraction $(1 - PY_h)$ of the utilized COD, the balance (PY_h) being converted into cell mass. Numerically with P = 1,48 mg COD.mg VSS^{-1} and Y_h = 0,45 mg VSS.mg COD^{-1}, the ratio (oxygen consumption) / (substrate utilization, expressed as COD) = 1/3. If the values of Y_h and P as found in aerobic systems apply also to anoxic/aerobic systems, it is possible to determine the ratio (nitrogen removal) / (substrate utilization) in a similar fashion: As 1 mg of NO_3-N is equivalent to 2,86 mg O_2 (Eq (2.26)) the utilization

of 1mg COD requires $(1-PY_h)/2,86$ = 0,117 mg NO_3-N. The utilization of organic material cannot be picked up experimentally from COD measurements as much of the biodegradable material is absorbed on the sludge, i.e. it is not part of the liquid phase.

3.5.4 Denitrification rate constants in plug flow reactors

In the investigation by Marais and his coworkers using pre- or postdenitrification plug flow reactors it was possible to analyse the kinetics of denitrification as a function of the actual retention time in the plug flow reactor. By sampling the nitrate concentration along the length of the plug flow reactors they obtained nitrate concentration profiles. Figs 2.16 and 2.17 show examples of these profiles measured by Stern and Marais (1974) and Marsden and Marais (1976) respectively.

Under constant flow and load conditions the nitrate concentration-retention time profiles typically exhibited shapes as indicated in Figs 2.18a and 2.18b, for pre- and postdenitrification reactors respectively. The postdenitrification profiles all exhibited a single phase linear decrease of the nitrate concentration with time, i.e. a behaviour that can be described by Eq (2.31). The predenitrification profiles however all indicated a two-phase denitrification response, i.e. (1) a fast primary phase of short duration (1-9 mins) followed by (2) a slower secondary phase that persisted for the balance of the time in the plug flow reactor. The slope in the secondary phase was steeper than in the single phase of the postdenitrification reactor.

From an analysis of the experimental data over a range of sludge ages (10 to 20 days) Marais and his coworkers came to the very important conclusion that the rates of denitrification are *not* proportional to the total *volatile* solids concentration X_v as suggested in Eq (2.31), but proportional to the *active* sludge concentration X_a. (The concentration of X_a was calculated from Eq (2.32)). Hence, they reformulated Eq (2.31),

$$(dN/dt) = - K'X_a \qquad (2.38a)$$

where

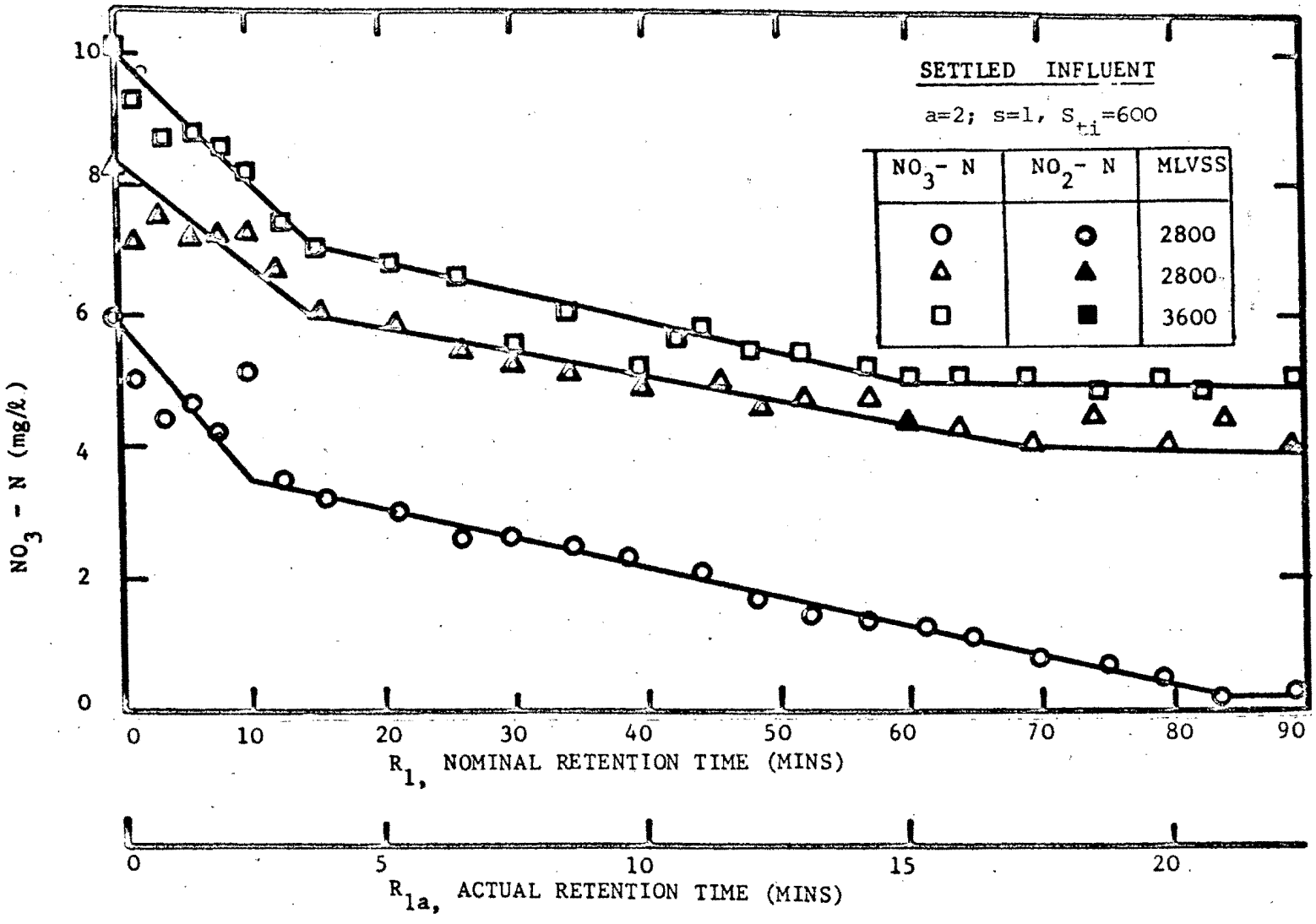

Fig.2.16 *Typical nitrate concentration - retention time profiles in a pre-denitrification reactor.*

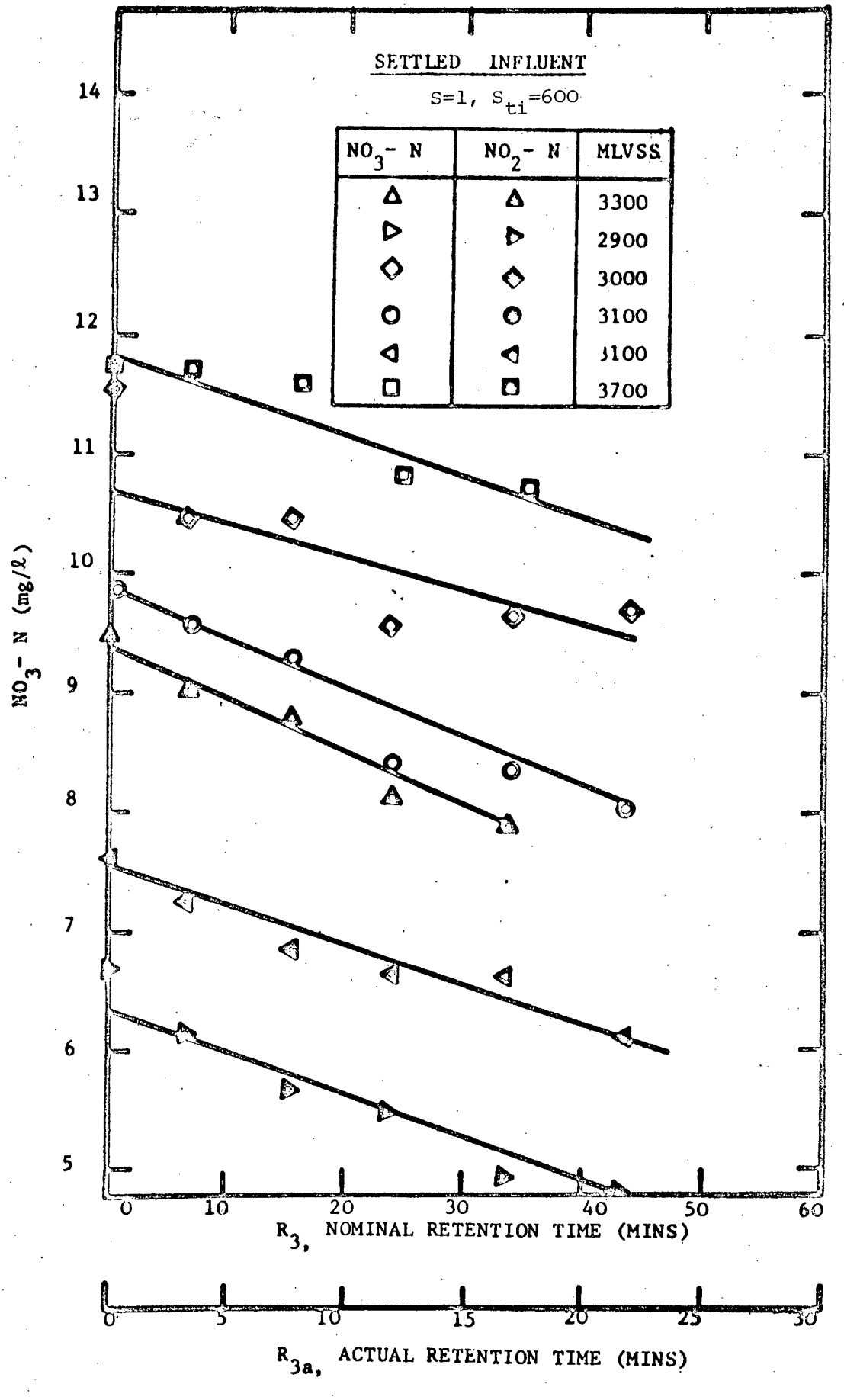

Fig. 2.17 *Typical nitrate concentration - retention time profiles in a post-denitrification reactor.*

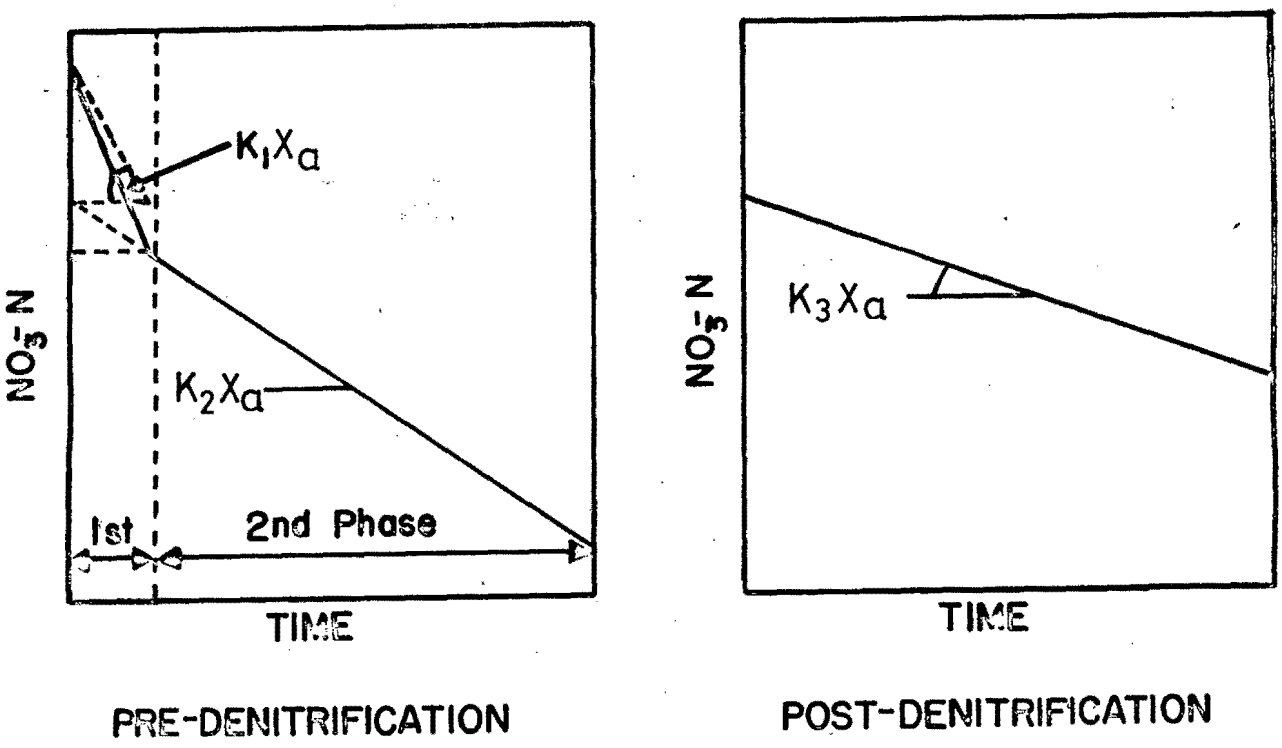

Fig.2.18(a and b) *Nitrate concentration - retention time profiles in anoxic plug flow reactors; pre-and post denitrification respectively.*

K' - denitrification rate constant (mg N.mg X_a^{-1}.time^{-1})

To describe denitrification in the predenitrification plug flow reactor,Stern and Marais accepted two denitrification rate constants: K_1' acting during the primary phase and K_2,acting during the secondary phase of denitrification. An alternative approach is to assume that in the primary phase the high denitrification rate is due to two simultaneous processes: a primary rate with a denitrification rate constant K_1 that persists only over the period of the primary phase, t_p, and a secondary rate with a denitrification rate constant K_2 that persists for the total time in the anoxic reactor (interrupted lines in Fig 2.18). These two approaches are equivalent: in the primary phase $K_1' = K_1 + K_2$. The alternative approach,however,has some calculation advantages. More important, it will be shown in Chapter 4 that for a mathematical description of activated sludge process kinetics,the alternative approach is basically more acceptable. The alternative approach will be used in the present work .

The relationships between the decrease of the nitrate concentration and the actual retention time in an anoxic plug flow

reactor can now be expressed as:

a) Predenitrification reactor

$$\Delta N_{1a} = (K_1 + K_2) X_a R_{1a} \qquad (R_{1a} < t_p) \tag{2.39}$$

or

$$\Delta N_{1a} = K_1 X_a t_p + K_2 X_a R_{1a} \qquad (R_{1a} > t_p)$$

$$\Delta = \Delta N'_{1a} + K_2 X_a R_{1a} \tag{2.40}$$

where

ΔN_{1a} = observed or *reactor* removal of nitrates in a predenitrification reactor (mg NO_3 - N.ℓ^{-1})

$\Delta N'_{1a}$ = *reactor* nitrate removal due to the denitrification process with rate K_1 (mg NO_3 - N.ℓ^{-1})

t_p = duration of the primary phase (h)

R_{1a} = actual retention time in the predenitrification reactor (h)

K_1, K_2 = denitrification rate constants in a predenitrification reactor (mg NO_3 - N.mg X_a^{-1}.h^{-1})

b) Postdenitrification reactor

$$\Delta N_{3a} = K_3 X_a R_{3a} \tag{2.41}$$

where

ΔN_{3a} = *reactor* nitrate removal in a postdenitrification reactor reactor (mg NO_3-N.ℓ^{-1})

K_3 = denitrification rate constant in a postdenitrification reactor (mg NO_3-N.mg X_a.h^{-1})

R_{3a} = actual retention time in the postdenitrification reactor (h)

Note that Eqs (2.39, 2.40 and 2.41) are only valid if the nitrate concentration does *not* become zero in the anoxic reactor. If complete denitrification takes place it is not possible to calculate K_1, K_2 or K_3 on the basis of these equations.

From numerous experimental profiles generated by Stern and Marais (1979) and Marsden and Marais (1976) the values of K_1, K_2 and K_3 were calculated. The experimentally determined values show a considerable spread, with K_1 and K_2 exhibiting log-normal distributions. Fig 2.19a shows statistical plots of K_1 and K_2 at 14°C and at 20°C calculated from experiments reported by Marsden and Marais (1976) and Stern and Marais (1974) respectively. Fig 2.19b shows a plot for K_3 at 14°C (after Marsden and Marais, 1976).

Statistical comparisons of the respective mean K_1, K_2 and K_3 values at the same temperature but at different sludge ages (within the investigated range of 10 to 20 days), influent COD concentrations (300 to 800 mg COD.ℓ^{-1}) or recycle ratio (1 to 10) did not show any significant trends and it was concluded that the denitrification rate constants are independent of sludge age, influent COD and recycle ratios.

Temperature dependencies of K_1, K_2 and K_3 can be determined from Figs. 2.19a and b. Assuming an Arrhenius type equation, the denitrification rate constants can be expressed as:

$$K_1 = 0{,}03(1{,}20)^{T-20} \quad \text{mg N.mg } X_a^{-1}.h^{-1} \qquad (2.42)$$

$$K_2 = 0{,}0042(1{,}08)^{T-20} \quad \text{mg N.mg } X_a^{-1}.h^{-1} \qquad (2.43)$$

$$K_3 = 0{,}0032(1{,}03)^{T-20} \quad \text{mg N.mg } X_a^{-1}.h^{-1} \text{ *} \qquad (2.44)$$

The values of K_1, K_2 and K_3 in Eqs (2.42 - 2.44) are mean values. The data from which these were determined showed wide

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* The value of K3 at 20°C was obtained from experiments with a completely mixed reactor.

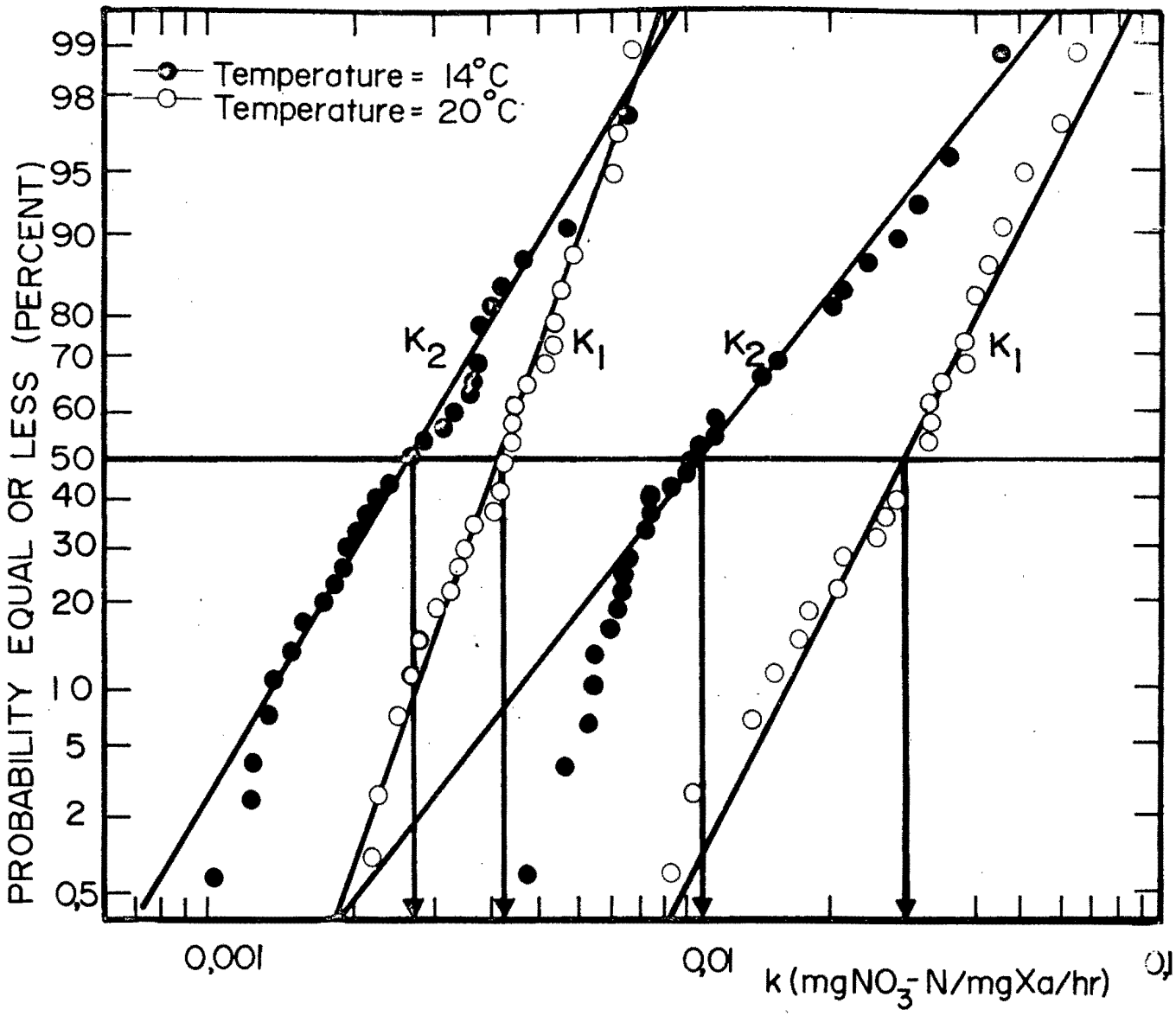

Fig.2.19a *Statistical plots of observed denitrification rate constants, K_1 and K_2, at 14°C(Marsden and Marais)and at 20°C(Stern and Marais).*

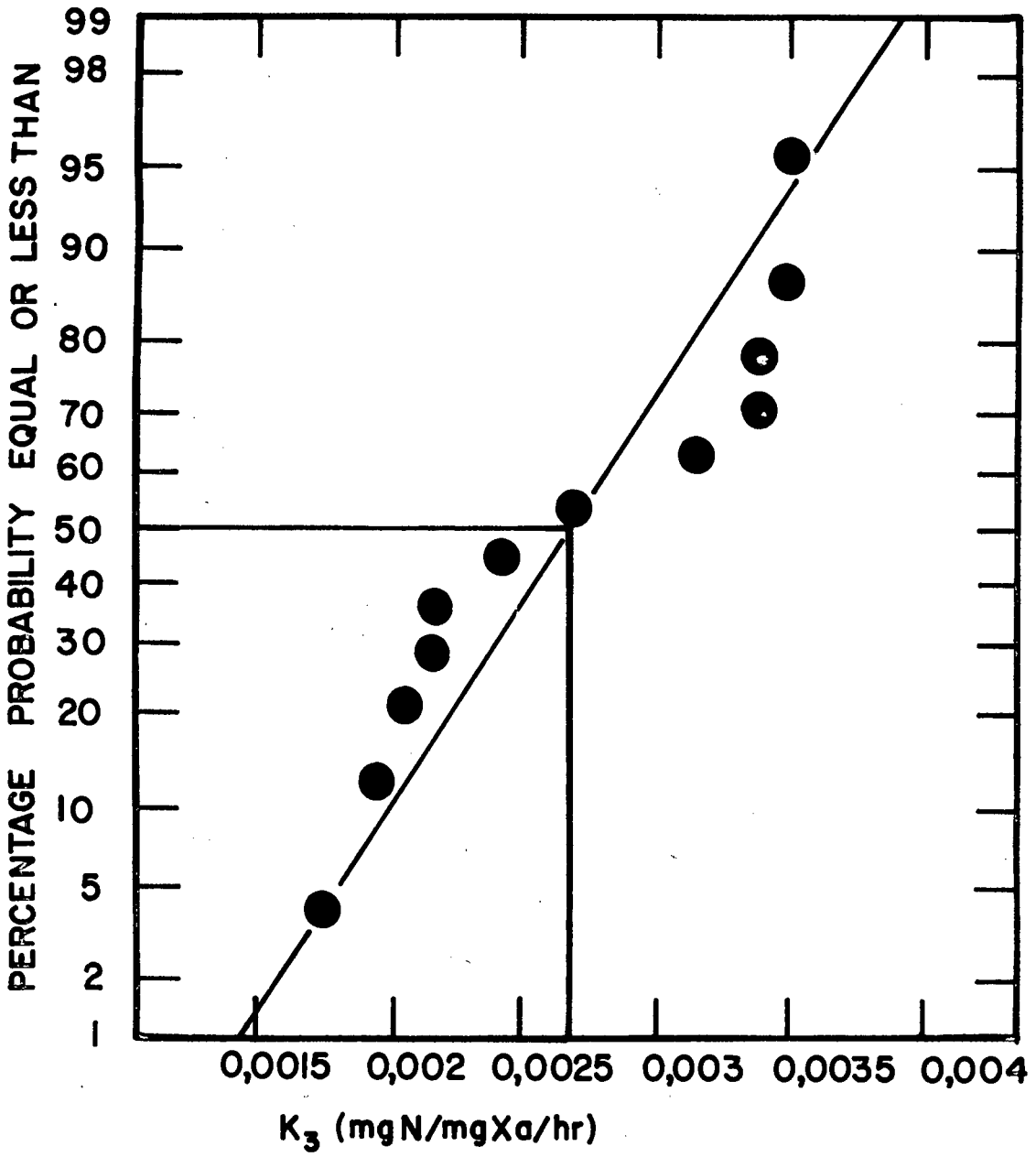

Fig.2.19b *Statistical plot of the observed denitrification rate constant in a post-denitrification reactor, K_3 at 14°C(Marsden and Marais).*

variation (see Figs 2.19a and b). Although the standard deviations of the means are small, it should not be inferred that the means have a corresponding stability; reviewing the data indicates that sequences of low and high values were often obtained. The lengths of these sequences were usually associated with the feed periods of batches of influent sewage (one batch was used for 1-2 weeks). Variation of $\pm$ 25 per cent in the values of K_1, K_2 and K_3 were not uncommon. Very evidently the changes in these values were in part due to changes in the sewage characteristics from batch to batch. Identification of some of the causes of the changes will become possible after modelling the general behaviour of the process, in Chapter 4.

3.5.5 Reactor nitrate removal and system nitrate removal

In denitrification calculations it is often convenient to express the nitrogen removal in terms of mg NO_3-N per litre *influent*, i.e. the *system* nitrate removal, ΔN_s, instead of mg NO_3-N per litre *mixed liquor*, i.e. the *reactor* nitrate removal, ΔN_a. In systems under constant flow and load conditions these two parameters are related. This can be shown as follows: In an anoxic reactor with volume V_{an}, if there is a decrease of the nitrate concentration ΔN_a due to denitrification then the rate of denitrification is given by

$$r_D = \Delta N_a / R_a \qquad (2.45)$$

where

R_a = actual retention time in the anoxic reactor. The actual retention time is given by the ratio of the volume V_{an} to the *actual* flow that passes through the reactor, i.e. the influent flow plus the recycle flows. With a denitrification rate r_D and a volume V_{an} the mass of nitrate removed in the anoxic reactor per time unit, ΔM_n, is given by:

$$\Delta M_n = r_D . V_{an} \qquad (2.46)$$

During a time unit, a volume Q (Q = influent flow rate) of sewage passes through the system so that the mass of nitrate removed per unit volume of influent, i.e. the *system* nitrate removal is given by:

$$\Delta N_s = \Delta M_n/Q = r_D V_{an}/Q = r_D R \qquad (2.47)$$

where

R = *nominal* retention time in the anoxic reactor (h or d)

The nominal retention time is equal to the ratio of the volume V and the influent flow Q. Comparing Eqs (2.45 and 2.47)

$$r_D = \frac{\Delta N_s}{R} = \frac{\Delta N_a}{R_a} \qquad (2.48)$$

or

$$\Delta N_s = \Delta N_a . R/R_a \qquad (2.49)$$

The ratio R/R_a is given by:

$$R/R_a = (V/Q)/(V/((a+s+1)Q)$$

$$= (a+s+1) \qquad (2.50)$$

where

a and s are recycle ratios to the anoxic reactor (Fig 2.9)

Hence, the ratio system:reactor nitrate removal is given by:

$$\Delta N_s/\Delta N_a = (a+s+1) \qquad (2.51)$$

Eq (2.40) shows that in a predenitrification reactor with a nominal retention time $R_1 > t_p(a+s+1)$ the system nitrate removal can be written as

$$\Delta N_{1s} = \Delta N'_{1s} + K_2 X_a R_1 \qquad (2.52)$$

where

$\Delta N'_{1s}$ = system nitrate removal due to process with rate constant K_1

From the analysis of the data reported by Marais and his coworkers it could be established that the system nitrate removal, N'_{1s} was independent of recycle ratios, temperature, sludge age and sludge concentration but proportional to the influent biodegradable COD, i.e.

$$\alpha = \Delta N'_{1s}/S_{bi} \qquad (2.53)$$

= proportionality constant between the system nitrate removal due to the denitrification process with high rate and the influent biodegradable material (mg NO_3 - N.mg COD^{-1})

In Fig 2.20 a statistical plot is shown plotted for a biodegradable influent COD of S_{bi} = 492 mg COD.ℓ^{-1}

(i.e. a total influent COD of S_{ti} = 600 mg COD.ℓ^{-1})

The average value of $\Delta N'_{1s}$ is 13,6 mg NO_3-N.ℓ^{-1} so that the value of α is calculated at:

α = 0,028 mg NO_3 - N.mg COD^{-1}

Hence in general terms the system nitrate removal in a predenitrification plug flow reactor is given by:

$$\Delta N_{1s} = \alpha S_{bi} + K_2 X_a R_1 \qquad (R_1 > t_p(a+s+1)) \qquad (2.54)$$

where R_1 = nominal retention time in the predenitrification reactor.

For a postdenitrification plug flow reactor from Eq (2.41)

with Eq (2.48)

$$\Delta N_{3s} = K_3 X_a R_3 \qquad (2.55)$$

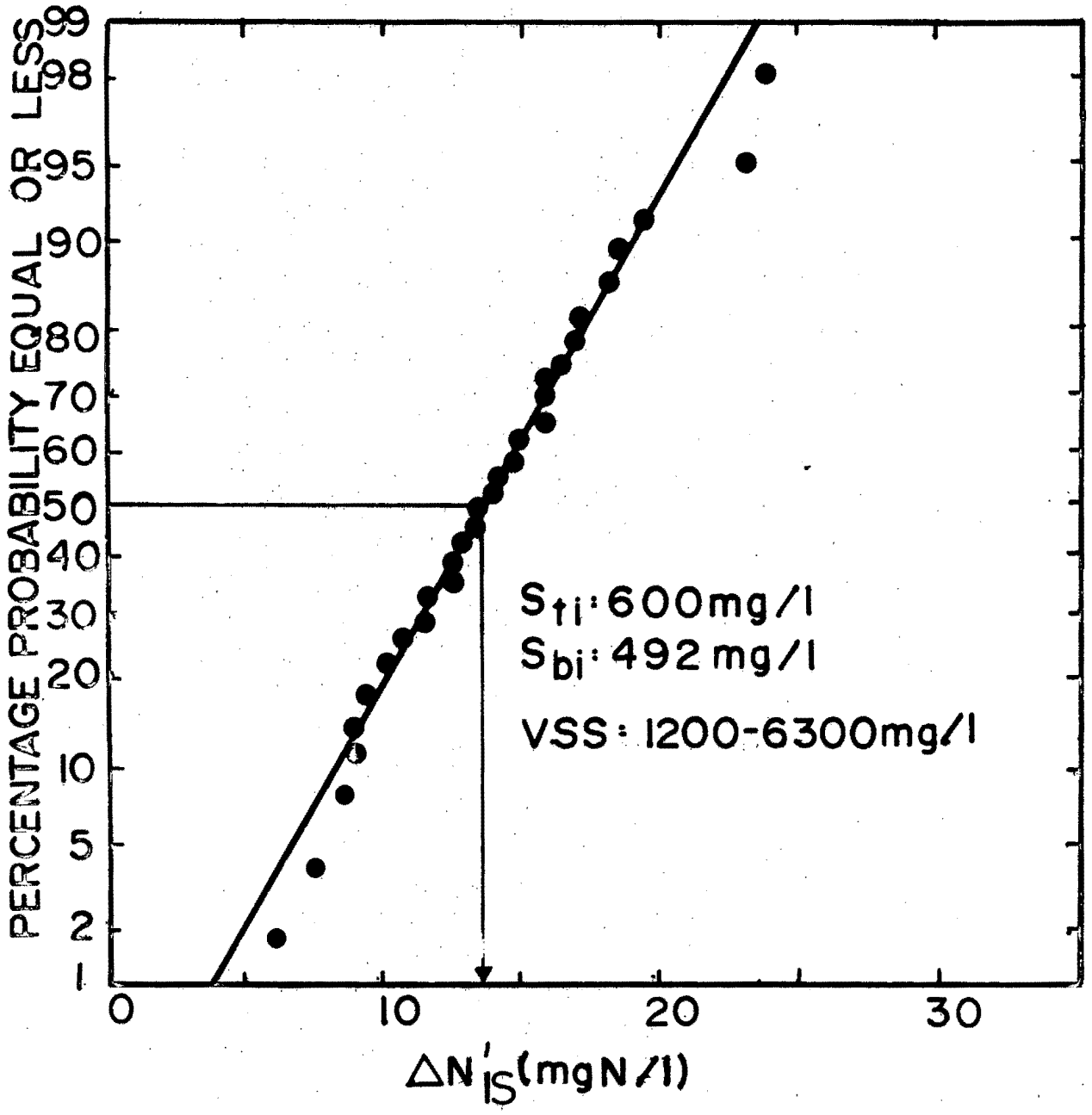

Fig. 2.20 *Statistical plot of the primary phase system nitrate removal in a pre-denitrification plug flow reactor.*

where

R_3 = nominal retention time in the postdenitrification reactor.

For a system with both a pre- and postdenitrification plug flow reactor one obtains:

$$\Delta N_s = \Delta N_{1s} + \Delta N_{3s}$$

$$= \alpha S_{bi} + K_2 X_a R_1 + K_3 X_a R_3 \qquad (2.56)$$

3.5.6 Denitrification in completely mixed reactors

3.5.6.1 Experimental calculations

Nitrate removal in completely mixed reactors can be calculated from the nitrate concentrations in the different reactors and in the settler and the recycle ratios. Consider a predenitrification system (Fig 2.9) with nitrate concentration N_{an} in the anoxic zone, N_{ae} in the aerobic zone and N_{se} in the settler (theoretically the latter two should be the same and equal to the effluent nitrate concentration as,ideally,no reaction takes place in the settler. However, in practice these values are often slightly different). Assuming recycle ratios "a" from the aerobic to the anoxic reactor and "s" from the settler to the anoxic reactor, the influent nitrate concentration entering the anoxic reactor with the influent and the recycles now can be calculated by taking the weighted average:

$$N_{in} = (N_i + aN_{ae} + sN_{se})/(a+s+1) \qquad (2.57)$$

where

N_{in} = composited nitrate concentration entering the anoxic zone.

N_i = nitrate concentration in the influent.

Usually $N_i = 0$ and

$$N_{in} = (aN_{ae} + sN_{se})/(a+s+1) \qquad (2.57a)$$

With a nitrate concentration N_{an} in the anoxic reactor (and hence

in the effluent from this reactor) the reactor nitrate removal is:

$$\Delta N_{1a} = (N_1 + aN_{ae} + sN_{se})/(a+s+1) - N_{an} \qquad (2.58)$$

The system nitrate removal is now given by Eq (2.49)

$$\Delta N_{1s} = N_i + aN_{ae} + sN_{se} - (a+s+1)N_{an} \qquad (2.59)$$

In a postdenitrification system (Fig 2.7) the calculation is somewhat simpler. The reactor nitrate removal is given by the difference between the nitrate concentrations in the aerobic and in the anoxic reactors.

$$\Delta N_{3a} = N_{ae} - N_{an} \qquad (2.60)$$

and the system nitrate removal is a factor (s+1) times greater, i.e.

$$\Delta N_{3s} = (s+1)(N_{ae} - N_{an}) \qquad (2.61)$$

A difficulty that sometimes arises is that in an anoxic reactor there is an increase of the nitrite concentration due to incomplete reduction of the nitrate ion. In this case an *equivalent* nitrate removal may be calculated. If the nitrate concentration in an anoxic reactor is reduced by ΔNO_3 and the nitrite concentration increased by ΔNO_2 (normally $\Delta NO_3 > \Delta NO_2$), then it can be assumed that a quantity $\Delta NO_3 - \Delta NO_2$ of nitrate is reduced to nitrogen gas and the remainder, i.e. ΔNO_2, to nitrite. The oxidation number of nitrate is 5, of nitrite +3 and of molecular nitrogen 0, i.e. two electrons are transferred in the reduction of nitrate to nitrite and 5 in the reduction of nitrite to nitrogen gas. Hence the reduction of 1 mole of nitrate to nitrite is, in oxidimetric terms, equivalent to the reduction of $\frac{2}{5} = 0,4$ mole of nitrate to nitrogen gas. Therefore, if in an anoxic reactor the nitrite concentration increases by ΔNO_2, this is equivalent to a reduction of nitrate to nitrogen gas of $0,4.\ \Delta NO_2$, so that in the above example the equivalent nitrate removal would be:

$$\Delta N_{1a} = \Delta NO_3 - \Delta NO_2 + 0,4\ \Delta NO_2 \qquad (2.62)$$

$$= \Delta NO_3 - 0,6\ \Delta NO_2$$

Eq. (2.62) can be written also in terms of the influent and effluent nitrate and nitrite concentrations of the anoxic reactor:

$$\Delta N_{1a} = NO_{3in} - NO_{3e} - NO_{2e} + NO_{2in} + 0,4(NO_{2e} - NO_{2in})$$

$$= (NO_{3in} + NO_{2in}) - (NO_{3e} + NO_{2e}) + 0,4(NO_{2e} - NO_{2in}) \qquad (2.63)$$

where index in refers to influent to the anoxic reactor

index e refers to effluent from the anoxic reactor

In practice the reactor nitrate removal is often taken to be equal to the difference in the sum of nitrate and nitrite concentrations in the influent to,and in the effluent from,an anoxic reactor. From Eq (2.63),if there is an increase in the nitrite concentration in the anoxic reactor this procedure is in error and the *equivalent* reactor nitrate removal is larger than the observed difference; for an increase if 1 mg NO_2-N.ℓ^{-1} in the anoxic zone there is an increase of 0,4 mg NO_3-N.ℓ^{-1} in the equivalent reactor nitrate removal.

It is also possible to calculate the experimental system nitrate removal on the basis of a *nitrogen* balance over the entire system. The influent nitrogen can be (1) discharged as TKN with the effluent, (2) wasted with the excess sludge, (3) nitrified and discharged as nitrate, and (4) nitrified and denitrified. Doing as mass balance over the system:

$$\Delta N_s = N_{ti} - N_{te} - N_{ne} - N_s \qquad (2.64)$$

where

ΔN_s = system nitrate removal

N_{ti} = influent TKN concentration

N_{te} = effluent TKN concentration

N_{ne} = effluent nitrate concentration

N_s = nitrogen wasted with sludge

(all as mg N.ℓ^{-1} influent)

The values of N_{ti}, N_{te}, and N_{ne} can be measured directly. In order to establish the value of N_s it can be assumed that the mass fraction of nitrogen in the wasted sludge is f_n = 0,1 mg N.mg VSS^{-1} and that a fraction $1/R_s$ of the total sludge mass MX_v is wasted per day. Hence the daily mass of nitrogen required for sludge wastage, MN_s, is:

$$MN_s = Q.N_s = f_n MX_v/R_s = f_n.V.X_v/R_s \quad (2.65)$$

so that

$$N_s = f_n X_v R_h/R_s \quad (2.66)$$

where

N_s = influent TKN concentration required for incorporation in wasted sludge

MN_s = daily required nitrogen for sludge wastage.

MX_v = total sludge mass.

Q = influent flow.

R_h = total hydraulic retention time.

R_s = sludge age.

V = total reactor volume.

X_v = VSS concentration.

If data for the sludge concentration, X_v, is not available it can be estimated by substituting for X_v from Eq (2.35) in Eq (2.66):

$$N_s = f_n(Y_h(1+fb_{hT}R_s)(1-f_{us}-P.f_{up})/(1+b_{hT}R_s) + f_{up})S_{ti} \quad (2.67)$$

A problem in the calculation of the nitrogen balance is that it is assumed that all the denitrification takes place in the anoxic reactor or reactors. In practice some denitrification may take place in the settler or in poorly aerated aerobic reactors; consequently, the system nitrate removal calculated from a nitrogen balance over the whole system may be expected to be slightly higher than the system nitrate removal calculated from a nitrate balance over the anoxic reactor.

3.5.6.2 Theoretical calculations

The fact that linear nitrate profiles are obtained in plug flow reactors (Fig 2.18) indicates that in the particular states present in these reactors the denitrification rate approximates a zero order reaction. Consequently, if completely mixed reactors replace the plug flow reactors, the nitrate removal, again should be amenable to estimation by means of Eq (2.56), which implicitly contains the zero order approximation. It was repeatedly found by Marais and his co-workers that in experiments where plug flow reactors were replaced by completely mixed reactors, the denitrification achieved in these reactors was, within experimental error, in accordance with the predictions of Eq (2.56) using the values K_1, K_2 and K_3 (Eqs 2.42 to 2.44)) and α (Eq (2.53)),determined experimentally from observations on anoxic plug flow reactors.

It is now interesting to analyze if the denitrification behaviour observed by other research workers can also be predicted in terms of Eq (2.56). Most reported work cannot be used as, almost invariably, insufficient data are given to calculate the active sludge concentration by means of Eq (2.32), so that it is not possible to check the validity of Eq (2.56). However, in the work presented by Sutton *et al* (1979) previously cited, all the required data for the calculation of X_a are available and an analysis of the theoretical denitrification behaviour can be made and compared with the reported observations. Sutton *et al* carried out denitrification experiments at pilot plant scale and using both pre- and postdenitrification systems with completely mixed reactors using raw municipal sewage. Only experiments where the nitrate

concentration in the anoxic reactor was higher than 0,5 mg NO_3-N.ℓ^{-1} are considered here, because it is possible that for lower values the nitrate concentration could be a factor limiting the denitrification rate.

a) Predenitrification systems

Predenitrification systems were operated at various temperatures, sludge ages and anoxic sludge mass fractions. A two reactor system i.e. an anoxic reactor followed by an aerobic reactor was used (Fig 2.9) with a 1:1 recycle ratio from the settler to the anoxic reactor and a 3:1 recycle ratio from the aerobic to the anoxic reactor. For the calculation of the theoretical system nitrate removal it is assumed that the concentration of X_a is given by Eq (2.32) i.e.

$$X_a = Y_h R_s (1-f_{us}-P.f_{up})\bar{S}_{ti}/[(1+b_{hT}R_s)R_h] \qquad (2.68)$$

where

$\bar{S}_{ti}$ = average influent COD concentration (mg COD.ℓ^{-1})

f_{us} = 0,11 (mg COD.mg COD^{-1})*

f_{up} = 0,17 (mg VSS.mg COD^{-1})* (see Fig 2.14)

The average influent COD concentration, $\bar{S}_{ti}$, calculated from all the influent COD concentrations reported by Sutton *et al*, was equal to 301 mg COD.ℓ^{-1}. The system nitrate removal due to the denitrification process with the high rate constant K_1, $\Delta N'_{ls}$, was calculated from the total influent COD concentration, S_{ti}, on the day the experiment was carried out i.e.

$$\Delta N'_{ls} = \alpha(1-f_{us}-P.f_{up})S_{ti}$$

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* The values for f_{us} and f_{up} were discussed previously : Section 3.5.2.

where

α = 0,028 mg NO_3 - N.mg S_{bi}^{-1} (Fig 2.20)

In Table 2.12 *all* the reported data of predenitrification systems at various temperatures and sludge ages with a nitrate concentration $N_{an} > 0,5$ mg N.ℓ^{-1} in the anoxic reactor are represented. The experimental system nitrate removal is calculated with Eq (2.59) and the theoretical system nitrate removal with Eq (2.56), using Eq (2.32) to determine X_a and Eq (2.53) to determine $\Delta N'_{is}$. From Table 2.12 it may be noted that although the observed and theoretical values may differ substantially in the individual experiments, the average observed system nitrate removal is very closely predicted. It may be noted that these data include temperature variations from 7 to 25°C and sludge ages from 3,5 to 20 days.

b) Post-denitrification systems

In Table 2.13 *all* the reported data of postdenitrification systems at various temperatures and sludge ages are presented. The experimental system nitrate removal is calculated using Eq (2.61) after correcting for nitrites using Eq (2.63). The theoretical system nitrate removal was calculated using Eq (2.55). From the data in Table 2.13 it is clear that there is close correlation between the experimental and theoretical values at 15°C and 25°C, but the theoretical system nitrate removal at 7 to 8°C consistently appears to be too high. The K_3 value used for the theoretical calculation was extrapolated using the Arrhenius relationship for K_3 (Eq (2.44)) determined for the range 14°C to 20°C. From the experimental data presented it would appear that the temperature dependency of K_3 in the region between 7 to 14°C is much stronger than in the region between 14 to 20°C: For the best correlation between experimental data and theoretical predictions at 7 to 8°C the denitrification rate constant is only K_3 = 0,0012 mg NO_3.mg X_a^{-1}.h^{-1} which is only a fraction 0,47 of the K_3 value at 14°C, implying in a temperature dependency coefficient of Θ = 1,12 for K_3 between 7 and 14°C as against a temperature dependency coefficient of Θ = 1,03 between 14°C and 20°C. It is possible that the K_3 value at 14°C itself is

Table 2.12 Experimental and theoretical nitrate removals in predenitrification systems (experimental data from Sutton et al, 1979)

Number	T°C	R_s (d)	R_h (h)	R_1 (h)	S_{ti}	NO_{3ef}	NO_{3an}	ΔN_{1s}	
								Exp	Theory
p1.1	15,0	6	7,0	2,0	298	11,4	6,1	15,1	15,0
p1.2	15,0	6	7,9	2,0	306	9,6	3,5	16,7	15,2
p2.47	7,5	20	8,75	2,5	230	7,3	3,0	11,3	14,1
p2.48	7,0	20	8,75	2,5	280	7,4	2,4	14,1	15,0
p2.49	7,0	20	8,75	2,5	260	8,2	2,8	15,0	14,8
p2.50	14,5	7	7,0	2,0	240	15,1	7,8	17,1	14,4
p2.51	15,0	7	10,0	6,67	220	19,5	8,8	24,0	14,1
p2.57	25,0	3.5	5,0	2,0	330	8,4	4,9	7,3	12,4
p2.58	22,0	3,5	8,75	2,0	368	6,2	2,9	8,4	11,9
							Mean	14,4	14,7

T = temperature (°C)

R_s = sludge age

S_{ti} = influent COD

NO_{3ef} = effluent nitrate concentration

NO_{3an} = nitrate concentration in the anoxic reactor

$(\Delta N_s)_{exp}$ = observed system nitrate removal (mg N.ℓ^{-1})

$(\Delta N_s)_{theor}$ = theoretical system nitrate removal (mg N.ℓ^{-1})

Recycle ratios : s = 1; a = 3.

Table 2.13 Experimental and theoretical nitrate removals in postdenitrification systems (experimental data from Sutton et al (1979).

Number	T(°C)	(R_s(d)	R_h(h)	R_3(h)	Aerobic		Effluent		ΔN_{3s}	
					NO_3	NO_2	NO_3	NO_2	Exp	Theory
p1.3	25	10	10,0	6,0	10,9	0,0	4,5	1,3	11,0	12,4
p1.4	26	10	10,0	6,0	10,3	0,1	3,9	0,3	12,4	12,4
p1.5	15	18	10,0	6,0	10,0	0,2	2,2	0,9	15,2	12,5
p1.8	15	12	10,0	6,0	8,6	0,0	3,8	0,9	8,4	11,3
p1.9	15	12	10,0	6,0	9,5	0,1	1,9	2,1	12,6	11,3
p1.9	14,5	7	10,0	6,0	10,0	0,9	6,5	1,4	6,4	9,5
p1.20	25	3	10,0	6,0	9,3	1,1	4,6	2,9	2,0	7,7
p1.21	25	3	10,0	5,0	8,5	2,0	4,5	4,0	5,6	6,3
p1.25	25	17	10,0	6,0	10,5	0,0	7,2	0,3	6,2	13,9
p1.26	25	16	10,0	6,0	11,0	0,0	5,0	0,2	12,0	13,0
p1.27	15	15	10,0	5,0	14,4	0,0	11,5	0,0	5,8	11,4
p2.51	15,5	18	10,0	6,7	10,1	0,1	0,2	0,4	19,8	17,8
p2.54	24	10	10,0	7,1	11,9	0,2	4,2	1,1	14,2	14,7
								Mean	10,5	11,8

Continued

Number	T(°C)	(R_s(d)	R_h(h)	R_3(h)	Aerobic		Effluent		ΔN_{3s}	
					NO_3	NO_2	NO_3	NO_2	Exp	Theory
p1.14	8	21	10,0	6,0	6,0	0,2	2,6	0,9	5,6	13,1
p1.15	7	21	10,0	6,0	4,8	0,2	1,7	0,7	3,8	13,1
p1.43	7,5	32	12,5	6,2	8,9	0,1	5,1	0,8	7,2	11,7
p1.44	7,5	35	12,5	7,5	7,9	0,8	0,2	0,3	16,0	14,3
p2.60	8	20	10,0	5,0	6,7	0,5	9,2	1,5	4,4	10,0
p2.61	8	20	10,0	5,0	5,5	0,9	3,7	1,6	3,2	10,0
								Mean	6,7	12,0

Recycle ratio : s = 1

For symbols see Table 2.12

overestimated but the data obtained on the K_3 value show such wide dispersion that a considerable amount of data must be collected to determine a reliable value. It is apparent that more data is needed in the lower temperature ranges before the temperature dependency is adequately established. It may also become necessary to reformulate the temperature dependency itself, as the simplified Arrhenius equation used at present is only empiric, useful over restricted temperature ranges. A third consideration is that in the lower temperature ranges below say 10°C, there is a qualitative change in the organism mass: cryophilic organisms may become dominant over mesophilic.
In all the calculations of the experimental system nitrate removal the *nitrate* balance over the anoxic reactor was used, i.e. it was calculated by Eq (2.59) and (2.61) for the pre- and postdenitrification reactor systems respectively. It was not possible to calculate the system nitrate removal on the basis of a nitrogen balance over the whole system because Sutton *et al* only reported the *filtered* influent TKN.

From their data Sutton *et al* came to the conclusion that all the denitrification rate constants, even though expressed as mg NO_3 - N.mg $X_a^{-1}.d^{-1}$, decreased with increasing sludge age. Their calculation, however, did not take account of the particulate unbiodegradable fraction, f_{up}, in the influent. From the analysis above it would appear that if the particulate unbiodegradable fraction, f_{up}, is taken into consideration, the denitrification rate constants do not vary with sludge age and in the temperature range from 15 to 25°C for both pre- and postdenitrification the values are very close to those calculated from the work of Marais *et al* (Eqs (2.42 to 2.44)).
In Fig 2.21 the ratios of (experimental system nitrate removal)/(theoretical system nitrate removal) are shown plotted as a function of the sludge age for the data presented in Tables 2.12 and 2.13. The data for postdenitrification at 7 - 8°C are not included.
Fig 2.21 shows that, while there is a considerable spread in the data, there is no distinct trend with sludge age. The average ratio $(\Delta N_{s,exp})/(\Delta N_{s,theor})$ is slightly less than 1,0 i.e. the theory tends to overpredict slightly the extent of denitrification.
Possible reasons for this small difference are:

(1) Dissolved oxygen introduced into the anoxic reactors with the recycle flows from the aerobic reactor and the settler; this would tend to reduce the extent of denitrification in the anoxic reactors.

(2) In predenitrification reactors only the sum of NO_3^- + NO_2^- concentrations in the anoxic and aerobic reactor is reported. It is possible that the nitrite concentration increased in the anoxic reactor and this would cause the experimental extent of denitrification to be slightly underestimated (Eq (2.63)).

(3) The α value for this waste flow may be smaller than that measured in South Africa.

Despite the small consistent deviation of the data by Sutton *et al* from those reported in South Africa, it seems that the calculation of nitrogen removal based on the empirical equations developed by Marais *et al* applies closely also to waste flows outside South Africa.

Analysis of the data supplied by Heideman (1979) in the fashion shown above was not possible because the anoxic sludge fraction with which he operated the system was so large ($^2/_3$ of the process volume) that complete denitrification in the anoxic reactor always took place. It was impossible, therefore, to do any kinetic analysis.

3.5.7 Minimum anoxic sludge mass fraction

In the calculations above it has been assumed tacitly that the retention time in the predenitrification reactor was sufficient for the process with the high denitrification rate constant K_1 to be completed. It will now be shown that this assumption was justified. The *system* nitrate removal due to high rate denitrification, i.e. $\Delta N'_{1s}$, is constant and equal to $\alpha.S_{bi}$ (Fig 2.20 and Eq (2.53)). Hence the reactor removal is given by:

$$\Delta N'_{1a} = \alpha . S_{bi}/(a+s+1) \qquad (2.70)$$

The minimum actual retention time for completion of the denitrification reaction with rate constant K_1 is given by (Fig 2.19a)

$$\Delta N'_{1a} = K_1 . X_a . t_p \qquad (2.71)$$

Now by equating Eqs (2.70 and 2.71) the minimum retention time t_p can be expressed as:

$$t_p = \alpha . S_{bi}/(a+s+1) \; /(K_1 . X_a) \qquad (2.72)$$

Note that, because the denitrification process with rate constant K_1 is a zero order reaction, this equation is valid for both completely mixed and for plug flow reactors. By substituting from Eq (2.38) in Eq (2.72) and reducing

$$t_p = (\alpha) \; /(a+s+1) \; . \; R_h/(K_1 . C_r) \qquad (2.73)$$

Knowing the actual retention time, t_p, the minimum nominal retention time to completion is given by:

$$R_{min} = t_p(a+s+1) \qquad (2.74)$$

The ratio of the minimal anoxic nominal retention time R_{min} to the total nominal retention time in the reactor, R_h, is given by:

$$R_{min}/R_h = V_{min}/Q/V/Q = V_{min}/V = f_{min} \qquad (2.75)$$

where

V_{min} = minimum reactor volume for completion of the process with high denitrification rate.

V = total reactor volume.

f_{min} = minimum anoxic volume or mass fraction* that allows completion of the process with high denitrification rate.

Substituting Eq (2.75) in Eq (2.73), the minimum anoxic sludge mass fraction, f_{min}, for completion of the process with rate constant K_1, is:

$$f_{min} = \alpha/(K_1 C_r) \qquad (2.76)$$

From Eq (2.76) it can be shown that f_{min} is very small so that under normal operational conditions (say $T > 14^{o}C$; $R_s > 10d$) the sludge mass fraction in a predenitrification reactor always will be much larger than f_{min}. Hence it can be assumed that the process with high denitrification rate constant K_1 is complete in the predenitrification reactor. In all the predenitrification systems operated by Sutton *et al* (Table 2.12) the anoxic sludge mass fraction, f_{xi}, was always much greater than the value for f_{min} calculated from Eq (2.76) and the system nitrate removal, therefore, could be calculated by means of Eq (2.54).

3.5.8 Denitrification potential

In Sections 3.5.5 and 3.5.6 above it was shown that the empirical model for denitrification, expressed by Eqs (2.54 to 2.56) is applicable to pre- and/or postdenitrification systems and to plug flow or completely mixed reactors under constant flow and load conditions. We will now investigate the factors that influence the maximum extent of denitrification in a particular anoxic/aerobic system under constant flow and load conditions.

For a predenitrification system:

$$\Delta N_{1s} = \Delta N'_{1s} + \Delta N''_{1s}$$
$$= \alpha S_{bi} + K_2 X_a R_1 \qquad (2.77)$$

* For a uniform sludge concentration throughout the system the sludge volume fraction in the anoxic reactor is equal to the sludge mass fraction in that reactor

where

R_1 = nominal retention time in the predenitrification reactor.

$\Delta N'_{1s}$ = system nitrate removal due to the process with rate constant K_1.

$\Delta N''_{1s}$ = system nitrate removal due to the process with rate constant K_2.

The value of N_{1s} is the *maximum* system nitrate removal attainable in a predenitrification reactor and for this reason is called the *denitrification potential* of the reactor, D_{p1}. Maximum removal can be attained only if enough nitrate is introduced into the reactor, i.e. if denitrification in the anoxic reactor is *not* complete. In many instances it is not possible to introduce enough nitrates to develop the denitrification potential. In this event the anoxic reactor is, in effect, too large. The denitrification potential therefore forms a criterion for optimization and is used extensively in this regard in Chapter 5. Because of their future importance the denitrification potentials for a pre- and a post denitrification reactor are developed below:

For the predenitrification reactor (Eq (2.54))

$$D_{p1} = \alpha.S_{bi} + K_2 X_a R_1 \qquad (2.78)$$

D_{p1} = denitrification potential of a predenitrification reactor

By substituting for X_a from Eq (2.32) and noting that R_1/R_h is equal to the sludge mass fraction in the predenitrification reactor, f_{x1}:

$$D_{p1} = \alpha S_{bi} + K_2 C_r S_{bi} R_1/R_h$$

$$= (\alpha + K_2 C_r f_{x1}) S_{bi} \qquad (2.79)$$

Similarly for a postdenitrification reactor

$$D_{p3} = K_3 X_a R_3$$

$$= K_3 C_r f_{x3} S_{bi} \qquad (2.80)$$

where

f_{x3} = sludge mass fraction in the postdenitrification reactor.

In a system with both a pre- and postdenitrification reactor

$$D_p = D_{p1} + D_{p3} = (\alpha + K_2 C_r f_{x1} + K_3 C_r f_{x3}) S_{bi} \qquad (2.81)$$

From Eq (2.39) in the special case that the predenitrification reactor is so small that the high rate denitrification process cannot be completed the denitrification potential is given by:

$$D_p = ((K_1 + K_2) C_r f_{x1} + K_3 C_r f_{x3}) S_{bi} \quad (f_{x1} < f_{min}) \qquad (2.81a)$$

From Eq (2.81) the denitrification potential or maximum system nitrate removal is directly proportional to the influent biodegradable COD and linear to the anoxic sludge fractions in the pre- and postdenitrification reactors. It depends further upon the temperature (which determines the value of the denitrification rate constants) and upon the sludge age ($C_r = Y_h R_s/(1+b_{hT} R_s)$).

3.5.9 Limitations of the empirical model

While the extent of denitrification in the experiments described in the previous sections is accurately predicted by the denitrification capacity, i.e. by using the experimentally determined constants α, K_2 and K_3, this approach is essentially empirical; no overt link with the basic theory of the activated sludge process is evident, other than that the rate of denitrification is linked to the concentration of *active* sludge (which is calculated from theory) rather than to the total volatile solids concentration. The denitrification rate constants were determined from observations on systems under constant flow and load conditions and apply only under these conditions: variations of the rate and the extent of denitrification occur under cyclic flow and load conditions as reported by Wilson and Marais (1976). No link could be established between the rates under cyclic flow and load conditions and those under

constant flow and load conditions. The applicability of the empirical model therefore is restricted to very specific conditions.

To develop a general model for the kinetics of denitrification, attention was directed to the general theory for the activated sludge process for aerobic systems developed by Dold, Ekama and Marais (1980), to inquire if it was possible to extend this theory to include denitrification. This approach did not seem unreasonable when considering that NO_3 and O_2 serve the same function - as the electron acceptor. In order to extend the aerobic model it is important that its basic kinetics are thoroughly appreciated. For this reason a critical exposition of the aerobic model is given in Chapter 3.

CHAPTER THREE

KINETICS OF THE AEROBIC ACTIVATED SLUDGE PROCESS

1. INTRODUCTION

One of the main objectives of wastewater treatment in the activated sludge process is the removal of organic material from the liquid phase. This removal takes place either by oxidation of the organic material or by incorporation into the sludge mass present in the activated sludge process. If organic material is oxidized it is transformed into stable mineral products (CO_2 and H_2O) that either remain in the liquid phase or escape to the atmosphere. If the organic material is incorporated in the sludge mass, the organic material is not destroyed but it can be separated from the liquid phase by gravity settling.

The ecology of the sludge mass in an activated sludge process is complex. Besides numerous species of single celled organisms - bacteria, fungi and protozoa - there is a variety of higher organisms. Despite the complex nature of the ecosystem , it would appear that the kinetic behaviour of the process insofar as the removal of organic material from waste streams is concerned, can be modelled as if the sludge mass were an equivalent bacterial suspension, to which is attributed the observed behaviour of the process. It must be emphasized that the equivalent suspension may exhibit characteristics quite different from one composed of a single bacteria species only.

Efforts at modelling the activated sludge process based on observed behaviour commenced with the work of Monod (1949). Monod related the specific growth rate of the organisms to the concentration of substrate(biodegradable material) surrounding the organism and proposed an empirical relationship to this effect known as

the Monod equation.* Monod also concluded that for a specific biomass and constant environmental condition the mass of newly generated organisms per unit of nutrient utilized remained approximately constant. The constant of proportionality was termed the growth yield coefficient.

Herbert (1958), in an attempt to account for the decrease of biomass when no nutrient was supplied, suggested that a maintenance energy requirement must be satisfied through endogenous metabolism; that is some of the bacterial mass was oxidized to obtain maintenance energy. The rate at which the mass loss took place (the endogenous respiration rate) was proportional to the concentration of biomass present and this proportionality constant has become known under a number of designations; the one we shall use is the endogenous respiration rate constant. McKinney (1962) and McKinney and Simons (1964) were the first investigators to show that during endogenous respiration not all the active biomass that disappears is utilized for energy: approximately 18 to 22 per cent remains as unbiodegradable endogenous residue. McCarty and Brodersen (1962) and Washington and Hetling (1964) investigated the endogenous residue intensively by noting the increase in VSS concentration in an aerobic reactor fed continuously with glucose but from which no sludge was wasted. Their work constituted the first positive proof that the endogenous residue existed.

All the theoretical work discussed above was based on the implicit assumption that the organic substrate was of such a nature that the organism could absorb it directly. A different formulation was developed by Andrews and his group (Blackwell, 1971; Andrews and Busby, 1973; Stenstrom, 1975). They suggested that storage of nutrient occurs prior to absorption and metabolic utilization. They proposed empirical

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* Penfold and Morris (1912) proposed a model that can be reduced to the same form as that of Monod but apparently it did not have the impact that Monod's had.

McKinney (1962) also proposed a relationship different from that of Monod which has been the cause of considerable controversy in activated sludge kinetics. Monod's approach has been accepted in preference to McKinney's, not because it necessarily leads to better predictions, but because Monod's equation can be related conceptually to enzyme kinetics.

formulations to describe the rates of absorption and storage and subsequent utilization respectively. Stenstrom made a particularly valuable contribution by proposing that the rate of utilization of stored material should be expressed in terms of (mass of stored material/mass of sludge) instead of material stored as a bulk concentration.

Dold, Ekama and Marais (1980) advanced the kinetic theory of adsorption, substrate utilization and sludge growth in three important aspects:

(1) They showed that Stenstrom's kinetic equation can be developed from the surface active site theory for surface chemical reactions proposed by Levenspiel (1972), provided the volatile solids concentration in Stenstrom's equation is replaced by the active mass concentration. By doing this they accorded an acceptable basis to the reaction in the same way that Michaelis-Menton enzyme kinetic formulation formed the basis for Monod's equation.

(2) They demonstrated that the biodegradable organic matter in normal municipal wastes consists of two fractions: a soluble one that reacts in accordance with Monod's model and a particulate or colloidal one, which behaves in accordance with the model based on Levenspiel kinetics.

(3) They questioned the synthesis-endogenous respiration approach, an approach that until that time had been used generally in kinetic models. It accordance with the endogenous respiration approach a maintenance energy is required during endogenous respiration, the energy being generated from the destruction of organism cell material. In their view,this led to inconsistencies in the physical interpretation of the process. Instead they proposed a model which in essence did not recognize an energy requirement for maintenance. They proposed that in the activated sludge process organisms die at a fixed rate; the dead organisms lyse their organic material back to the surrounding medium where part of it is resynthesized (regenerated) by the live organisms into new organism mass. They concluded that the average expectancy of life of the microorganisms was approximately 1,1 days. Consequently in an activated sludge process with a

solids retention time longer than 1 to 2 days the continuous regeneration of organisms is such that the constitution of the heterotrophic organism mass does not change with increasing solid retention time (sludge age).*

Incorporation of these three aspects into the kinetics of the activated sludge process formed the basis of their bisubstrate-active site-death regeneration model.

In the development of the kinetic relationships between substrate utilization and sludge growth, sanitary engineers have always been confronted with the problem of how to express the "organic pollution strength" of the influent. Whereas bacteriologists, using well-defined organic media, can express the concentration of organic material in terms of chemical composition of the media, this is not possible in waste water technology because of the highly complex mixture of organic materials. Around the turn of the century a parameter was proposed by the Royal Commission for Sewage Treatment and Disposal - the 5 day Biochemical Oxygen Demand - BOD_5. This parameter in time attained a dominant position for defining the organic pollution strength. Although the BOD_5 has performed a useful function for many purposes, particularly in the practical control of full scale plants, its use in the development of the kinetic theory of the activated sludge process has not been so successful. The reason for this is that the test reflects only the organic material actually oxidized in the metabolic activities of the organism mass in the sample, not the total carbonaceous material content in the sample. A test of later origin, called the Chemical Oxygen Demand, does reflect the total organic content, but has not attained great popularity in kinetic descriptions of the activated sludge process because it does not differentiate between biodegradable and unbiodegradable material in the influent. Marais

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* Dold, Ekama and Marais verified this conclusion by carrying out aerobic batch digestion tests on samples of mixed liquor taken from activated sludge processes. The logarithmic decrement of the oxygen uptake rate from one day to the next in the batch digestion tests of up to 10 days remained constant and did not vary irrespective of whether the sample was taken from a process having a short sludge age (2,5 days) or a long sludge age (30 days).

and Ekama (1976) however demonstrated the advantages in using the COD rather than the BOD_5 concentration as a parameter in describing the process response: Utilizing the COD as a parameter they were able to perform a mass balance of the carbonaceous material in an activated sludge process under constant flow and load conditions, by equating the daily mass of COD entering the activated sludge system to the sum of the daily mass of COD in the effluent, the mass of COD wasted as sludge and the daily mass of oxygen utilized for carbonaceous material oxidation; the last term, by definition, being equal to the mass of COD oxidized per day. Such a mass balance is not possible when the BOD_5 concentration is used as a parameter (see Section 2).

In all the work done prior to that of Marais and his group, a general approach in testing the applicability of a kinetic hypothesis was to take the effluent quality as the criterion for comparison. It is clear that once adsoprtion and storage are accepted the effluent quality becomes a very poor parameter, not related in a straightforward manner to organism metabolism. For this reason Marais and his group suggested the oxygen uptake rate as a superior parameter as it is directly related to the metabolic activity of the organism mass. In all their work the oxygen uptake rate became the principle parameter in terms of which the predictions of any proposed theory was evaluated. The use of this parameter, in conjunction with the COD, proved so successful that it is probably true to say that without these two parameters their model could not have been developed.

Marais and Ekama (1976), having accepted the COD concentration and oxygen uptake rate as the basic parameters for organic pollution strength and biological activity respectively, and by calibrating their kinetic model against experimentally observed data under steady state operation found that the influent COD needed to be subdivided into the biodegradable and unbiodegradable materials, the unbiodegradable material being present as soluble and particulate fractions respectively. Later, Dold, Ekama and Marais (1980) on the basis of experimental results obtained on cyclicly loaded processes subdivided the biodegradable material also into two fractions: easily biodegradable and slowly biodegradable.

Having subdivided the COD into its various fractions, it was then found possible to distinguish three different categories of organic solid material that are derived directly or indirectly from the influent organic material:*

(1) active sludge derived from the biodegradable influent material,

(2) endogenous residue internally generated as the inert residue of the process of endogenous respiration.

(3) inert organic sludge derived from the unbiodegradable particulate matter in the influent.

The short review above briefly relates the historical development of theories on the kinetics of degradation and utilization of organic material relevant to the activated sludge process. Of these the bisubstrate-active site-death regeneration theory proposed by Dold *et al* probably gives the most realistic description available today of the kinetics of the aerobic activated sludge process. This theory will form the basis for extension into denitrification kinetics for single sludge activated sludge systems. In order to extend the theory it is important that a detailed understanding of its basic tenets is obtained. The remainder of this chapter will be concerned with describing the various aspects of the theory in detail.

2. ORGANIC MATERIAL IN WASTEWATER

The presence of biodegradable material in water is most promir tly characterised by its propensity to serve as an electron donor in an environment containing heterotrophic microorganisms and a suitable electron acceptor. Empirical methods have been devised to estimate the potential of electron transfer of organic material in waste water. Two tests to evaluate this potential that have found application in waste-

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* Later when adsoprtion and storage were recognised, a fourth sludge fraction was distinguished - the stored biodegradable material.

water technology are the COD and the BOD_5 tests (Standard Methods, 1971). In both the COD and the BOD_5 tests, the concentration of organic material is measured in terms of the concentration of oxidant (electron acceptor) that is required for the transfer of electrons from the organic material. The difference between the tests is that in the COD test the oxidant (dicromate-sulphuric acid) is so strong that the transfer of electrons is maximum, i.e., the oxidaton of organic material is complete or virtually complete, while in the biodegradable BOD_5 test only a part of the organic material is oxidized.

From the half reaction for the reduction of oxygen:

$$4e + 4H^+ + O_2 \rightarrow 2H_2O \qquad (3.1)$$

it may be noted that oxidimetrically 1 mol of molecular oxygen (32g) is equivalent to 4 electron equivalents. By definition, when the equivalent of 1g oxygen is used up in the COD test then the mass of COD oxidized is also 1g. Hence,

$$1g\ COD \doteq 1g\ Oxygen \doteq 1/8\ electron\ equivalent \qquad (3.2)$$

In the COD test, because organic material is oxidized completely, under constant flow and load conditions it is possible to perform a mass balance on an activated sludge process in terms of this parameter. The mass of organic material, as COD, entering the activated sludge process duly leaves the process as follows:

(1) A fraction leaves the system in the effluent flow, unmodified by the system.

(2) A fraction is destroyed by oxidation, i.e., electrons are transferred from the organic material to the electron acceptor. The extent of electron transfer (or COD destroyed) can be calculated from the measured oxygen uptake for carbonaceous energy degradation using Eq (3.2).

(3) A fraction is incorporated in the sludge mass by synthesis, adsorption and physical entrapment and leaves the system in the daily wasting of the sludge.

Thus the COD balance for the system can be written as:

$$MCOD_{inf} = MCOD_{eff} + MCOD_{sl} + MCOD_{ox} \quad (3.3)$$

where

$MCOD_{inf}$ = mass of COD in the influent per day

$MCOD_{eff}$ = mass of COD in the effluent per day

$MCOD_{sl}$ = mass of COD in wasted sludge per day

$MCOD_{ox}$ = mass of COD oxidized or oxygen utilized per day.

The validity of Eq (3.3) has been demonstrated by Marais and Ekama (1976) by obtaining COD recoveries of 96 per cent and better with respect to the influent COD. Eq (3.3) is particularly valuable in that it provides the research worker with a means to verify the correctness of his analytical procedures. In this respect the COD has an important advantage over the BOD_5: The influent BOD_5 value reflects only the oxygen utilized by the organisms in 5 days; some biodegradable material synthesized is not yet oxidized after 5 days of incubation and the inert organic material in the influent is not measured. The same applies to the effluent; the BOD_5 of the sludge wasted per day does not reflect the significant organic fraction of the sludge present as endogenous and inert materials - it is thus clear that utilizing the BOD_5 it is not possible to perform a mass balance on the process.

The COD balance (Eq (3.3)) can be rewritten in a more convenient form: Eckenfelder and Weston (1956) found that there exists a proportional relationship between the COD concentration of a sludge sample, (expressed as mg volatile solids per liter) i.e.,

$$COD_{ss} = P.X_v$$

COD_{ss} = COD concentration of a sludge sample.

X_v = volatile solids concentration of the sludge sample.

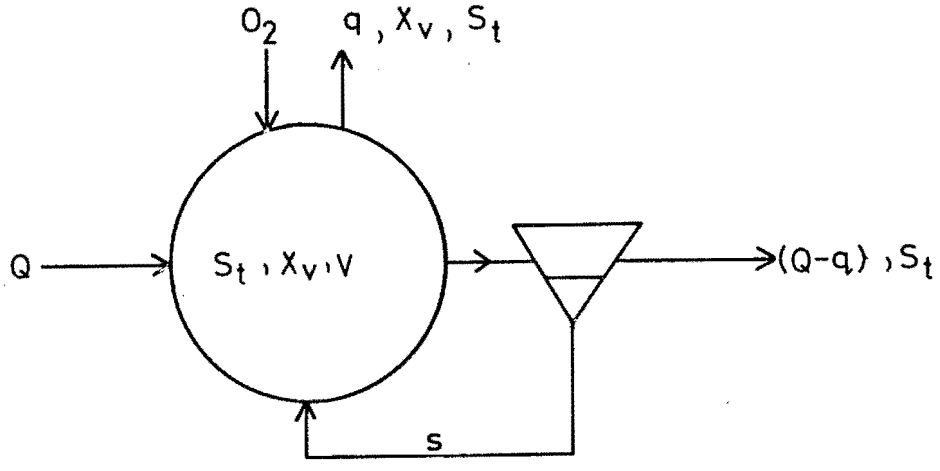

Fig. 3.1 *Completely mixed aerobic activated sludge process.*

P = proportionality constant

= COD/VSS ratio.

Reported values for P have ranged from 1,42 to 1,60 mg COD.mg VSS^{-1}. Marais and Ekama (1976) after an extensive investigation on processes with sludge ages ranging from 10 to 30 days reported an average value for P = 1,48 mg COD.mg VSS^{-1}. The use of this value appears to result in good COD balances.*

Once the P value of sludge from a particular process is known or estimated the COD of an activated sludge sample can be calculated directly from the concentration of volatile solids. In this case the mass balance

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* Later experiments by Dold, Ekama and Marais (1980) appear to indicate that P might not be a constant as generally accepted. Variation from the value found by Marais and Ekama, however, will not result in significant changes in the constants utilized in the model.

(Eq (3.3)) can be rewritten as:

$$MCOD_{inf} = COD_{eff} + P.\Delta MX_v + MCOD_{ox} \quad (3.4)$$

where

ΔMX_v = mass of sludge wasted daily.

Other forms of the mass balance equation can be derived: From Fig. 3.1 with an influent flow, Q, a sludge waste flow, q, and a reactor volume, V, the hydraulic retention time, R_h, and the solids retention time or sludge age, R_s, are respectively:

$$R_h = V/Q \quad (3.5)$$

$$R_s = V/q \quad (3.6)$$

Hence Eq (3.4) can be rewritten as:

$$Q.S_{ti} = (Q-q)\ S_t + q\ (PX_v+S_t) + V.O_c$$

$$= Q.S_t + q.PX_v + VO_c \quad (3.7)$$

$$\text{i.e. } MS_{ti} = MS_t + P.\Delta MX_v + MO_c \quad (3.8)$$

where

S_{ti} = influent COD concentration (mg COD.ℓ^{-1})

S_t = effluent COD concentration (mg COD.ℓ^{-1})

X_v = concentration of volatile solids in the process (mg VSS.ℓ^{-1})

O_c = oxygen uptake rate for carbonaceous material degradation (mg O.$\ell^{-1}d^{-1}$)

MS_{ti} = mass of influent COD per day (mg COD d^{-1})

M_{st} = mass of effluent COD per day (mg COD d^{-1})

MO_c = mass of oxygen utilized for carbonaceous material degradation (mg O.d^{-1})

To express the mass balance in terms of concentrations, divide Eq (3.7) by Q and substitute from Eqs. (3.5 and 3.6).

Noting that $q/Q = R_h/R_s$,

$$S_{ti} = S_t + R_h.P.X_v/R_s + R_h.O_c. \qquad (3.9)$$

3. KINETICS OF ORGANIC SUBSTATE UTILIZATION AND OF HETEROTROPHIC SLUDGE GROWTH.

3.1 Influent organic material

In order to describe the kinetics of utilization of organic material by heterotrophic organisms in the activated sludge process, it was necessary to divide the influent organic material into different fractions. The need for this division arose during the development of the general model in order to obtain correspondence between the predicted and observed response of a process. The division into the various fractions was obtained by interactive development between the predictions of the model and experimental observations; it attained validity by the closeness of the correspondence when tested over a wide spectrum of process configurations, sludge ages and loading conditions. Necessarily the division will depend upon the model describing the activated sludge process and the fractions have meaning only in this context. In the model developed by Dold, Ekama and Marais (1980) four different influent fractions are distinguished:

(1) Unbiodegradable soluble volatile material.

(2) Unbiodegradable particulate volatile material.

(3) Easily biodegradable material.

(4) Slowly biodegradable material.

3.1.1 Soluble unbiodegradable volatile fraction

This fraction consists of organic material not affected by the biochemical or flocculation action of the sludge. It is discharged unmodified with the effluent. The fraction, f_{us}, can vary from values as low as 5 per cent of the total influent COD, S_{ti}, (Marais and Ekama, 1976) to values as high as 10 to 12 per cent. The higher fraction seems to be more general and is supported by the experimental results observed by a number of investigations in other countries. A filtered effluent COD concentration of 10 to 12 per cent of the influent COD has been reported by Heide, (Holland) 1977, Heideman, (USA) 1979, Sutton *et al*, (Canada) 1979. A fraction f_{us} = 0,10 mg COD. mg COD^{-1} is probably not unreasonable as an average value to be expected for municipal waste waters having a large domestic component.

3.1.2 Particulate unbiodegradable volatile fraction

This fraction is enmeshed in the sludge (bioflocculation) but does not undergo any chemical transformation. It is removed from the system in the sludge wasted. By calibration, using their model, Dold *et al* found that the unbiodegradable particulate fraction, expressed as COD, is about 13 per cent of the influent COD concentration, S_{ti}, in unsettled municipal sewage and 3 to 7 per cent in settled municipal sewage. Assuming a COD/VSS ratio of P = 1,48 mg COD. mg VSS^{-1}, numerically the inert volatile solids fraction, f_{up}, in the influent is given by:

f_{up} = 0,13/1,48 = 0,09 mg VSS.mg COD^{-1} (raw sewage)

f_{up} = (0,03 to 0,07)/1,48 = 0,02 to 0,05 mg VSS.mg COD^{-1} (settled sewage)

These values apply to South Africa; in Chapter 2 it was shown that the magnitude of f_{up} might be influenced strongly by various factors e.g. where garbage grinding with disposal to the sewers is practised.

3.1.3 Easily biodegradable fraction

This fraction is of a soluble nature and appears to be composed of organic molecules that can pass directly through the cell walls of microorganisms and react in a manner similar to glucose. Dold *et al* (1980) have estimated this fraction ,f_{ca}, at about 24 per cent of the biodegradable influent COD, S_{bi}, (f_{ca} = 0,24 mg COD.mg COD^{-1}) or, equivalently, about 19 per cent of the total influent COD, S_{ti}, for unsettled minicipal waste flows in South Africa. This fraction can vary considerably even in the same waste flow source. The reason for this is not clear; possibly, because of its readily available biodegradable nature it is utilized preferentially in long sewer lines by microorganisms attached to the sewer walls.

3.1.4 Slowly biodegradable fraction

This fraction is of a "particulate" nature and is assumed to be composed or large molecules, colloids and solid particles. These cannot pass directly through the cell wall. Instead the particulate biodegradable material is adsorbed and stored on the surface of active cells, and the stored material is then hydrolyzed by exoenzymes to smaller molecules prior to passing through the cell wall.

Summarizing the concentration of organic material in a wasteflow of domestic origin can be subdivided as follows:

$$S_{ui} = f_{us} \cdot S_{ti} \tag{3.10}$$

$$X_{ii} = f_{up} \cdot S_{ti} \tag{3.11}$$

$$S_{bi} = (1-f_{us}-P.f_{up})S_{ti} \tag{3.12}$$

$$S_{bsi} = f_{ca}.S_{bi} = f_{ca}.(1-f_{us}-P.f_{up}).S_{ti} \tag{3.13}$$

$$S_{bpi} = (1-f_{ca})S_{bi} = (1-f_{ca})(1-f_{us}-P.f_{up})S_{ti} \tag{3.14}$$

where

S_{ti} = total influent COD (mg COD.ℓ^{-1})

S_{bi} = biodegradable influent COD (mg COD.ℓ^{-1})

S_{ui} = soluble unbiodegradable influent COD (mg COD.ℓ^{-1})

X_{ii} = particulate unbiodegradable influent COD (mg VSS.ℓ^{-1})

S_{bsi} = easily biodegradable influent COD (mg COD.ℓ^{-1})

S_{bpi} = slowly biodegradable influent COD (mg COD.ℓ^{-1})

f_{ca} = ratio easily biodegradable : biodegradable COD

f_{us} = ratio soluble unbiodegradable : total COD

f_{up} = ratio particulate material : total COD

P = COD/VSS ratio = 1,48 mg COD.mg VSS^{-1}.

3.2 Composition of sludge in the mixed liquor

Sludge in the activated sludge process is composed of the settleable material in the mixed liquor. The total concentration of suspended solids (TSS) of a mixed liquor sample can be determined by standard procedures of liquid-solid separation and weighing after drying (Standard Methods, 1971). A fraction of the solids is inorganic and can be determined from the mass remaining after ignition of a dried TSS sample at 600°C and is called Inorganic Suspended Solids (ISS). The fraction lost on ignition at 600°C is principally organic material and is called Volatile Suspended Solids (VSS). For sludge derived from raw domestic sewage the VSS concentration is usually 75 to 80 per cent of the TSS concentration. The VSS is composed of live and dead solid organic material, the former comprising a great variety of microorganisms.

In the model of Dold *et al* (1980) the organic sludge mass is divided into five fractions. As in the case of the division of the influent COD into different fractions, the division of the sludge into the five fractions is defined in terms of the model and found by calibration of the predictions of the model against observed process behaviour. The theoretical fractions depend upon the influent sewage characteristics (the division of the influent organic material, S_{ti}, in its fractions S_{ui}, X_{ii}, S_{bsi}, and S_{bpi}) and operational conditions of the activated sludge process (temperature, sludge age). The proportions of the different sludge fractions are determined by fitting the theoretical response of the activated sludge process against experimentally observed data.

Dold *et al* distinguished the following sludge fractions:

(1) Active sludge mass

The active sludge mass consists of the live microorganisms retained in the process. The active mass is generated from the influent biodegradable material.

(2) Endogenous residue mass

In the active sludge process the live cells that form the active sludge mass die off at a constant rate depending on the temperature. Part of the dead cells is not biodegradable and of a particulate nature; i.e., it accumulates in the activated sludge process as endogenous residue.

(3) Inert organic mass

A fraction of the influent organic material is unbiodegradable and of a particulate nature. In the process this material is flocculated or enmeshed in the sludge mass, i.e., it becomes part of the solid phase.

(4) Stored organic material

Depending upon operational conditions in the activated sludge process a fraction of the slowly biodegradable influent material may be

stored on the active mass, i.e., it forms part of the solid phase but has not yet been utilized by the active mass.

(5) Autotrophic sludge mass

In addition to the volatile solids that arise directly or indirectly from the influent organic material there may also be a sludge mass of autotrophic organisms that develops in the process of nitrification.

3.3. Utilization of biodegradable organic material

3.3.1 Utilization of easily biodegradable material

Although it is likely that the fraction of easily biodegradable material in the influent sewage is composed of a large number of different substrates for the purpose of modelling it is taken as a mono-substrate and its rate of utilization is described by the Monod equation:

$$r_{us} = -(dS_{bs}/dt)_u = K_{msT} S_{bs}/(S_{bs}+K_{ssT})\ X_a \qquad (3.15)$$

where

r_{us} = rate of utilization of easily biodegradable material (mg COD.ℓ^{-1}.d^{-1})

s_{bs} = concentration of easily biodegradable material (mg COD.ℓ^{-1})

X_a = concentration of active sludge (mg VSS.ℓ^{-1})

K_{msT} = specific utilization rate constant (mg COD.mg VSS^{-1}.d^{-1})

K_{ssT} = half saturation constant (mg COD.ℓ^{-1})

Index "u" refers to utilization of substrate.

3.3.2 Utilization of slowly biodegradable material

Slowly biodegradable material undergoes two sequential reactions: (1) Adsorption and (2) Utilization.

(1) Adsorption

The process of adsorption can be formulated on mass transfer principles; the higher the mass adsorbed the smaller the driving force for adsorption and the lower the rate of adsorption, the higher the concentration of adsorbate (the biodegradable particulate material) the higher the rate of adsoprtion. Also there is a limit to the mass of adsorbate that can be adsorbed. Let the maximum mass of adsorbed (and consequently stored) material per unit of active mass be defined by:

$$F_{ma} = X_{s,max}/X_a \qquad (3.16)$$

where

F_{ma} = ratio of (maximum concentration of stored biodegradable material/ concentration of active mass)

$X_{s,max}$ = maximum concentration of stored material (mg VSS.ℓ^{-1})*

X_a = concentration of active mass (mg VSS.ℓ^{-1})

From adsorption kinetics it is possible to express the rate of adsorption as:

$$r_a = -(dS_{bp}/dt)_a = (dX_s/dt)_a/P = K_{vT}S_{bp}(F_{ma}-X_s/X_a)X_a \qquad (3.17)$$

where

r_a = adsorption rate (mg COD.ℓ^{-1}.d^{-1})

S_{bp} = concentration of biodegradable particulate material (mg COD.ℓ^{-1})

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* Note that the concentration of stored material is expressed as mg VSS.ℓ^{-1}

X_s = concentration of stored biodegradable material (mg VSS.ℓ^{-1})

K_{vT} = specific adsorption rate constant (ℓ.mg $VSS^{-1}.d^{-1}$)

P = COD/VSS ratio

$(F_{ma}-X_s/X_a)$ - driving force for adsorption

index "a" refers to adsorption.

(2) Utilization

Utilization of stored organic material requires two steps: (a) The stored material is hydrolyzed and, (b) the hydrolized material passes through the cell wall and is utilized for metabolic activities. The rate of hydrolysis of the stored material is hypothesized to be the rate limiting step. As storage is a necessary prerequisite for hydrolysis and for metabolism of slowly biodegradable material, the rate of utilization of stored material X_s may be expected to depend on the concentration of stored material <u>at the surface</u> of the active mass. For this reason Dold *et al* (1980) suggested to formulate the hydrolysis and synthesis reaction of stored material in terms of the kinetic expression proposed by Levenspiel (1972) for surface mediated reactions. In Levelspiel's equation the reaction rate at an active surface is determined by the number of adsorption sites ("active sites") and can be expressed as follows:

$$r = K_a.C_{as}/(K_b+C_{as}) \qquad (3.18a)$$

where

r = reaction rate

K_a, K_b = reaction constants

C_{as} = concentration of the reactant <u>at the active surface</u>

Dold *et al* (1980) suggested that the ratio X_s/X_a i.e., the concentration of stored material per unit mass of active sludge be taken as the relevant parameter for concentration of stored material (the reactant)

thus giving quantitative expression to the idea that the hydrolysis and utilization of stored material occurs at the surface of the active sludge mass; in the bulk of the mixed liquor no stored material is present. Hence formulation for the hydrolysis reaction of stored material becomes:

$$r_h = -P(dX_s/dt)_h$$

$$= [K_{mpT}P(X_s/X_a)/(K_{spT}+PX_s/X_a)]\ X_a$$

$$= [K_{mpT}PX_s/(K_{spT}X_a+PX_s)]\ X_a$$

where

r_h = rate of hydrolysis of stored material (mg COD.ℓ^{-1}.d^{-1})

K_{mpT} = specific rate constant for hydrolysis (mg COD.mg X_a.d^{-1})

K_{spT} = half saturation value (mg COD.mg X_a^{-1})

Index "h" refers to hydrolysis.

Once the hydrolysis step is completed and the hydrolized material has passed through the cell wall, it can be utilized in the metabolic activity of the organism; the latter step appears to be very much faster than the former, so that at any time the mass of hydrolyzed material will be very small and the rate of utilization of hydrolyzed material is limited to the rate of hydrolysis itself,* i.e.,

$$r_{up} = r_h = [K_{mpT}PX_s/(K_{spT}X_a+PX_s)]\ X_a \qquad (3.19)$$

where

r_{up} = rate of utilization of slowly biodegradable particulate material (mg COD.ℓ^{-1}.d^{-1}).

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* There has been no specific enquiry to substantiate this statement, except that the model response based on this hypothesis appears to give a good account of the kinetic behaviour observed.

In Eq (3.14) the constant K_{mpT} can be interpreted as the substrate utilization rate constant for stored biodegradable material.

3.4 Synthesis of active mass

It is assumed that the mass of active volatile material synthesized is proportional to the mass of COD utilized irrespective of the source of COD, i.e.,

$$(dX_a)_g = -Y_h(dS)_u \qquad (3.20)$$

$(dX_a)_g$ = increase of the active mass concentration due to synthesis (mg VSS.ℓ^{-1})

$(dS)_u$ = decrease of organic substrate concentration due to utilization by the active mass (mg COD.ℓ^{-1})

Y_h = growth yield coefficient for active organisms (mg VSS.mg COD^{-1})

= increase of active mass per unit mass of COD utilized.

The rate of growth of active mass depends upon the rates of utilization of the two substrates: easily biodegradable and stored material. It is hypothesized that these rates are independent of each other, i.e., that the total rate of substrate utilization is given by the sum of the expressions for the rate of utilization of easily biodegradable material (eq (3.15)) plus that for slowly biodegradable (stored) material (Eq (3.19)). The only reason for accepting this hypothesis is that by its use the correspondence between predicted response from simulation and observed response is very close. There are indications however, that when the active mass is placed in an environment extremely rich in both substrates the numerical values of the rate constants defining the specific rate of utilization of the two substrates have to be reduced to obtain good correspondence, i.e., the one reaction seems to be affected by the presence of the other. This has been observed in the contact reactor of the contact-stabilization process (Alexander, Ekama and Marais, 1979). However in all the other processes the independence of the two reactions appears to be justified. Hence:

$$r_g = (dX_a/dt)_g - Y_h(dS/dt)_u = Y_h(r_{us}+r_{up}) \quad (3.21)$$

where

r_g = rate of growth of active cells (mg VSS.ℓ^{-1}.d^{-1})

S = concentration of biodegradable substrate (mg COD.ℓ^{-1})

X_a = concentration of active cells (mg VSS.ℓ^{-1})

Y_h = yield coefficient for active organisms

= increase of active mass due to utilization of a mass unit of biodegradable substrate (mg VSS.mg COD^{-1})

r_{us} = rate of utilization of easily biodegradable material (mg COD.ℓ^{-1}.d^{-1}) (Eq (3.15))

r_{up} = rate of utilization of slowly biodegradable material (mg COD.ℓ^{-1}.d^{-1}) (Eq (3.19))

Index "g" refers to growth of active mass

Index "u" refers to utilization of organic substrate.

3.5 Endogenous mass loss

If no substrate is fed to the activated sludge and aeration is continued a decrease of the mass of sludge is observed. This loss of sludge mass can be modelled in two different ways, by (a) the endogenous respiration approach and (b) the death-regeneration approach.

(a) Endogenous respiration approach

In the endogenous respiration approach it is considered that the live mass needs energy for cell maintenance and that in the absence of an external substrate the cell mass is oxidized and mineralized, i.e, the active mass decreases.

This approach in fact describes a nett effect, it is a 'black box'

approach which gives no indication of the possible ways that endogenous respiration is achieved by the organisms. For example, it could be hypothesized that along with external storage of substrate (X_s) there is an internal storage inside the cells of food, to be used when no external substrate is available. (The externally stored material is rapidly depleted). Internal storage is unlikely if one considers the following: An activated sludge plant with a sludge age of 20 d shows only a 5 per cent nett growth of active mass per day. If organic substrate supply to the organisms were not limited the active sludge mass could probably double its mass in a few hours - the supply of substrate in the normal activated sludge process severely limits the growth rate of the live organisms; in fact the organisms are maintained in a highly stressed condition and competition for organic substrate will be intense. Under such conditions internal storage is very unlikely. Accepting the endogenous respiration approach the rate of active mass loss can be described as a first order reaction:

$$r_{er} = -(dX_a/dt)_{er} = b_{hT}X_a \qquad (3.22)$$

where

r_{er} = endogenous respiration rate (mg VSS.$\ell^{-1}d^{-1}$)

b_{hT} = endogenous respiration rate constant (mg VSS.mg $VSS^{-1}d^{-1}$)

Index "er" refers to endogenous respiration.

From the work of McCarty and Brodersen (1962), McKinney and Ooten (1964) and Washington and Hetling (1964) it is evident that during endogenous respiration not all the active sludge mass is biodegradable; a fraction of the active mass that disappears per se remains as particulate volatile material called the endogenous residue.*

The rate of formation of endogenous residue can be expressed as:

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* From observation in batch digestion of active sludge, as there is no increase of filtered COD with time, it may be concluded that all the endogenous residue is of a particulate nature (Marais and Ekama, 1976).

$$r_{fe} = (dX_e/dt)_{fe} = -f(dX_a/dt)_{er} = fb_{hT}X_a \qquad (3.23)$$

r_{fe} = rate of formation of endogenous residue (mg VSS.ℓ^{-1}.d^{-1})

X_e = concentration of endogenous residue.

f = fraction of active mass that is not biodegradable (mg VSS.mgVSS^{-1})

Index "fe" refers to the formation of endogenous residue.

The value of the fraction f has been found to vary between 0,18 and 0,22. Marais and Ekama (1976) found f = 0,20 to give good correspondence between simulated and observed response. It seems that this fraction is not significantly influenced by temperature.

The value of the endogenous respiration rate constant, b_{hT}, in principle can be determined from the decrease of the concentration of volatile suspended solids with time in a batch digestion test (Marais and Ekama, 1976). There are, however, three problems that reduce the applicability of this procedure: (1) Usually the initial proportion of active and non-active sludge in the sample to be digested aerobically is not known and must be determined by curve fitting. (2) In samples taken from processes at long sludge ages the relative decrease of the VSS concentration is small so that a small error in a VSS determination is reflected in a large error in the b_{hT} value. (3) The VSS test is destructive so that a large initial sample is required to carry out a long term batch digestion test.

Marais and Ekama (1976) suggested a different method for determining b_{hT}: In a batch digestion test they observed the value of the oxygen uptake rate as a function of time, making due corrections for nitrification to obtain the uptake rate for the carbonaceous degradation. The logarithm of the value for the corrected oxygen uptake rate was plotted versus aeration time to give a straight line with slope b_{hT}. The theory of this procedure is developed as follows:

From

$$(dX_a/dt)_{er} = -b_{hT}X_a$$

for a batch test:

$$X_{at} = X_{ao} e^{-b_{hT} t} \quad (3.24)$$

where

X_{ao} = initial active mass concentration (mg VSS.ℓ^{-1})

X_{at} = active mass concentration after time t (mg VSS.ℓ^{-1})

t = time of batch digestion (d)

The oxygen uptake rate is incorporated as follows: The disappearance of 1 mg active mass results in the formation of f mg of inert residue and hence oxidation of the remainder, (1-f). The mass of (1-f) mg active sludge has a COD equivalent of P(1-f) and this COD equivalent is oxidized. Hence the oxygen requirement per mg active mass that disappears *per se* is given by P(1-f) i.e.,

$$(dO/dt)_{er} = -P(1-f)(dX_a/dt)_{er} = P(1-f) b_{hT} X_{ao} e^{-b_{hT} t} \quad (3.25)$$

where

$(dO/dt)_{er}$ = oxygen uptake rate for endogenous respiration.

In Eq (3.25) taking Log:

$$\log_e (dO/dt)_{er} = \log_e (P(1-f)(b_{hT} X_{ao}) - b_{hT} t$$

$$= C - b_{hT} t \quad (3.26)$$

which defines the linear relationship between log (oxygen uptake rate) and aeration time with slope b_{hT}. Note that this approach bypasses the effect of inactive volatile solids from any source in the sample to be digested.

Using the approach described above, Marais and Ekama (1976) found that $b_{h20} = 0{,}24\ d^{-1}$ at 20°C. They also found that this value is independent of the sludge age of the activated sludge process from which the sample for batch digestion was obtained. From measurements at 14°C Marais and Ekama related the value of the endogenous respiration

rate constant to temperature.*

$$b_{hT} = b_{h20}\theta^{(T-20)} \qquad (3.27)$$

$$= 0,24\ (1,029)^{T-20}$$

(b) Death regeneration approach

In the endogenous respiration approach it was mentioned that one of the difficulties in visualizing this approach is that internal storage of biodegradable organic material in the cells is unlikely. If internal storage is unlikely, the question arises: how does the active organism mass survive? Alternatively, if it is hypothesized that the organism mass utilizes its mass to obtain energy then one would expect that the organism characteristics should show an 'aging' effect, yet the b_{hT} values determined by means of the oxygen uptake rate procedure indicate no change in b_{hT} for samples taken from processes with sludge ages ranging from 2,5 to 30 days or, in the batch test itself with time.

The considerations above led Dold, Ekama and Marais (1980) to propose a different model for the phenomenon of 'endogenous mass loss': that of death and regeneration. In the death-regeneration approach the active cells die off at a fixed rate, i.e., cease to exist as live cells and through a process of lysis a fraction of the organic material of the dead cell mass becomes available in particulate**

** From the experimental work of Marais and Ekama (1976) it was not possible to assign the nature of the biodegradable material lysed from dead cells, i.e. whether it was soluble easily biodegradable or particulate slowly biodegradable. In Chapter 4 it will be shown that on the basis of observed denitrification behaviour it is necessary to assume that the lysed material is of a particulate biodegradable material.

* A widely used method of determining b_{hT} is by plotting the inverse of the sludge age (1/sludge age) versus the load factor. This approach underestimates the value of b_{hT} by at least 50 per cent (Marais and Ekama, 1976).

biodegradable material to the live cells. The live cells synthesize this biodegradable material to new cell mass, i.e., regeneration of active sludge takes place. The fraction of the dead cell mass that is not regenerated remains as unbiodegradable particulate volatile material - the endogenous residue.

The death rate can be formulated as a first order reaction, i.e.,

$$r_d = -(dX_a/dt)_d = b'_{hT}X_a \quad (3.28)$$

r_d = death rate of active cells (mg VSS.ℓ^{-1}.d^{-1})

b'_{hT} = death rate constant (mg VSS.mg VSS.ℓ^{-1}.d^{-1})

Index "d" refers to death of active cells

The rate of formation of endogenous residue is now given as:

$$r_e = (dX_e/dt)_e = -f'(dX_a/dt)_d = f'b'_{hT}X_a \quad (3.29)$$

where

r_e = rate of formation of endogenous residue (mg VSS.ℓ^{-1}.d^{-1})

f' = unbiodegradable fraction of active cells in the death-regeneration approach (mg VSS.mg VSS^{-1})

Index "e" refers to the formation of endogenous residue.

The biodegradable material released through lysis becomes available at the following rate:

$$r_p = P(1-f')r_d$$

$$= P(1-f')b'_{hT}X_a \quad (3.30)$$

where

r_p = rate of internal generation of biodegradable material (mg COD.ℓ^{-1}.d^{-1})

The generated biodegradable material is utilized by the active sludge, the rate of regeneration of active sludge being given by:

$$r_r = Y_h r_p = Y_h P(1-f')b'_{hT}X_a \quad (3.31)$$

r_r = rate of regeneration of active sludge (mg VSS.ℓ^{-1}.d^{-1})

The nett rate of decrease of active sludge is the difference between the death rate and the regeneration rate.

$$r_{dr} = -(dX_a/dt)_{dr}$$

$$= r_d - r_r$$

$$= b'_{hT}X_a - Y_h P(1-f')b'_{hT}X_a$$

$$= [b'_{hT}(1-Y_h P(1-f'))]X_a \quad (3.32)$$

where

r_{dr} = nett rate of decrease of the concentration of active sludge

Index "dr" refers to death and regeneration.

Both the endogenous respiration and the death-regeneration approaches must describe the same rate of decrease of active mass concentration, rate of increase of endogenous residue and rate of oxygen uptake in batch digestion tests. Hence

$$r_{er} = r_{dr}$$

$$= b_{hT}X_a$$

$$= [b'_{hT}(1-PY_h(1-f'))]X_a \quad (3.33)$$

i.e.,

$$b_{hT} = b'_{hT}(1-PY_h(1-f')) \quad (3.34)$$

Also

$$r_{fe} = r_e$$

$$= fb_{hT}X_a$$

$$= f'b'_{hT}X_a \qquad (3.35)$$

i.e.,

$$f = f'b'_{hT}/b_{hT} \qquad (3.36)$$

From Eqs (3.34 and 3.36) another relationship between these constants can be derived that will be useful later:

$$b'_{hT}(1-f') = b_{hT}(1-f)/(1-PY_h) \qquad (3.37)$$

Having established the interrelationships between the constants of the endogenous respiration and death-regeneration approaches, the mathematical description of the rates of sludge mass loss and of oxygen consumption for oxidation of sludge (see next section) give identical results. This being so there would seem to be little point in using the more complicated death-regeneration approach to describe the loss of active mass and the formation of endogenous residue. However, other considerations would seem to indicate that the death regeneration approach gives a better physical description of these processes.

(1) From Eqs (3.34 and 3.36) for f = 0,2 and b_{h20} = 0,24 d^{-1} one obtains f' = 0,08 and b'_{hT} = 0,62 d^{-1}. Considering the values of f and f', from discussions with bacteriologists at the University of Cape Town, a value of f' = 0,08 for the unbiodegradable fraction of an organism is more acceptable than f = 0,2.

(2) Considering b'_{hT} and b_{hT} a value of b'_{hT} indicates that the half life of the organism mass is 1,1 day. This implies that for any sludge age greater than 1,1 day the heterotrophic organism mass remains at a mean age of 1,1 day, i.e., no 'aging effect' of the active sludge activity will be noticeable between sludge ages of say 2,5 and 30 days and this is indeed so from the results of batch sludge digestions reported by Marais and Ekama - the b_{hT} value calculated from the oxygen

uptake rates remained constant.*

(3) The death regeneration approach provides a simple solution to the problem of describing the active mass behaviour when no electron acceptor (O_2 or NO_3^-) is available (anaerobic environment). The organisms continue to die say at the same rate, as under oxygenated conditions, lyse their nutrient to the surrounding medium and, when oxygenated will exert a higher oxygen demand than that attributable to the influent organic material accumulated during the period of anaerobiosis. This, in fact, was repeatedly observed and is described functionally and accurately in terms of the death regeneration approach. Such a description is not possible with the endogenous respiration approach except if a qualitative change in the behaviour pattern is hypothesized.

The discussion above raises the question: Should the formulation of the process behaviour be completely in terms of the death regeneration approach? This is not always necessary; the steady state solutions for the concentrations of active and endogenous residue and for the oxygen uptake rate are the same, numerically, for either approach. It is only when cyclic flow conditions or anaerobic-anoxic-aerobic processes are dealt with that the death regeneration approach becomes essential to describe the process response accurately.

In the present work, expressions for processes under constant flow and load will be given in terms of the endogenous respiration approach because this approach is widely understood and gives the same numerical values as the death regeneration expressions. All work connected with the general model describing the activated sludge process in series reactors under variable flow and loan conditions is in terms of the death regeneration approach.

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* It may be argued that the synthesis endogenous respiration approach also will be supported by this argument; for $b_h = 0{,}24$ the half life is 39 days. However, if batch tests are carried out them, in terms of the death regeneration approach, aging should become apparent. In making a choice between the two approaches, (2) above by itself is probably not a sufficient argument. The points expressed in (1) and (3) are more decisive.

3.6 Oxygen uptake rate for synthesis of carbonaceous material

From the growth rate of active sludge (eq (3.21)) the rate of COD utilization for synthesis is given by:

$$r_{syn} = P(dX_a/dt)_g = PY_h(dS_b/dt)_u$$

where

r_{syn} = rate of COD utilization for synthesis (mg VSS.$\ell^{-1}d^{-1}$)

$(dS_b/dt)_u$ = the rate of COD utilization for both synthesis and oxidation (mg COD. $^{-1}d^{-1}$)

Hence the difference between $(dS_b/dt)_u$ and r_{syn} is the rate of oxidation of COD. As 1 mg COD $\equiv$ 1 mg O_2,

$$r_o = (dO/dt)_c = (dS/dt)_u - PY_h(dS/dt)_u$$

$$= (1-PY_h)(dS/dt)_u$$

r_o = oxygen uptake rate for carbonaceous material oxidation (mg O.$\ell^{-1}d^{-1}$)

0 = concentration of dissolved oxygen (mg O.ℓ^{-1})

index "c" refers to oxidation of carbonaceous material.

If both easily and slowly biodegradable material are present, the total rate of substrate utilization is given by the sum of the rates of utilization of both substrates, r_{us} and r_{up}, respectively:

Hence

$$r_o = (1-PY_h)(dS_b/dt)_u = (1-PY_h)(r_{us}+r_{up})$$

3.7 Basic differential equations for carbonaceous material degradation and heterotrophic sludge growth

In Sections 3.3 to 3.6 the following basic mechanisms have been identified:

(1) Utilization of easily biodegradable material:

$$r_{us} = [K_{msT} S_{bs}/(S_{bs}+K_{ssT})]\ X_a \qquad (3.15)$$

(2) Adsorption of slowly biodegradable material:

$$r_a = [K_{vT} S_{bp} (F_{ma} - X_s/X_a)]\ X_a \qquad (3.17)$$

(3) Hydrolysis and utilization of stored material:

$$r_{up} = [K_{mpT} P X_s/(P X_s + K_{spT} X_a)]\ X_a \qquad (3.19)$$

(4) Active sludge growth:

$$r_g = [Y_h (K_{msT} S_{bs}/(S_{bs}+K_{ssT}) + K_{mp} P X_s/(P X_s + K_{spT} X_a))]\ X_a \qquad (3.21)$$

(5) Death of active sludge:

$$r_d = [b'_{hT}]\ X_a \qquad (3.28)$$

Using the basic mechanisms listed above, the reaction rates of the parameters involved in the biological metabolism of the sludge mass can be developed:

Easily biodegradable material:

$$r_{sbs} = (dS_{bs}/dt)_r = -r_{us}$$

$$= -[K_{msT} S_{bs}/(S_{bs}+K_{ssT})]\ X_a \qquad (3.36)$$

Slowly biodegradable material:

$$r_{sbp} = (dS_{bp}/dt)_r = -r_a + P(1-f')r_d$$

$$= -[K_{vT}S_{bp}(F_{ma}-X_s/X_a) - P(1-f')b'_{hT}]\ X_a \qquad (3.37)$$

Stored material:

$$r_{xs} = (dX_s/dt)_r = (r_a-r_{up})/P$$

$$= [K_{vT}S_{bp}(F_{ma}-X_s)/P - K_{mpT}X_s/(X_sP+K_{spT}X_a)]\ X_a \qquad (3.38)$$

Active mass:

$$r_{xa} = (dX_a/dt)_r = Y_h(r_{us}+r_{up}) = r_d$$

$$= [Y_h(K_{msT}S_{bs}/(S_{bs}+K_{ssT}) + PK_{mpT}X_s/(X_sP+K_{spT}X_a)) - b'_{hT}]\ X_a \qquad (3.39)$$

Endogenous residue:

$$r_{xe} = (dX_e/dt)_r = f'r_d$$

$$= [f'b'_{hT}]\ X_a \qquad (3.40)$$

Oxygen uptake rate:

$$O_c = (dO/dt)_r = (1-PY_h)(r_{us}+r_{up})$$

$$= [(1-PY_h)\ K_{msT}S_{bs}/(S_{bs}+K_{ssT}) + K_{mpT}X_sP/(X_sP+K_{spT}X_a)]\ X_a \qquad (3.41)$$

Note that in the above expressions the differential quotients (i.e., dC/dt, where C is any one of the variables (S_{bs}, S_{bp}, X_s, X_a, X_e or O) refer only to the rate of change in concentration due to reaction, as indicated by the index r. In an activated sludge process the actual rate of change of the variables is composed of a reaction effect and a hydraulic effect, the latter depending on the influent and recycle flow pattern, the concentrations of these variables in these flows, the mixing regime and the reactor configuration. The process

equations will be presented in Section 9 for the single reactor process under constant flow and load conditions, and in Section 10 for the series reactor process under cyclically varying flow and load conditions.

Fig 3.2 gives a schematic representation of the metabolism of biodegradable material and of sludge growth and death in terms of the bisubstrate-active site-death regeneration model developed by Dold *et al* (1980).

4. NITROGENOUS MATERIAL IN WASTEWATER AND ORGANIC SLUDGE

4.1 Nitrogenous material in wastewater

Nitrogen in wastewater is present principally as free and saline ammonia (NH_3 and NH_4) and organic nitrogen (mainly urea and amino components. These substances constitute the Total Kjeldahl Nitrogen - TKN concentration. Oxidized nitrogen compounds (i.e., nitrate or nitrite) usually are absent in domestic wastewaters.

In the TKN test a sample is first digested, i.e., any organic nitrogen is converted to ammonia. Thereafter the ammonia concentration is measured by titrimetic or colorimetic methods (Standard Methods, 1971). Omitting the digestion step, the concentration of ammonia only is determined. The difference between the TKN and the ammonia concentrations gives the organic nitrogen concentration. The concentration of nitrite is determined by colorimetic procedures (Standard Methods, 1971). Nitrate is reduced to nitrite and before its concentration can be determined by the same colorimetic procedures.

4.2 Nitrogen in organic sludge

Marais and Ekama (1976) determined the mass ratio TKN:VSS of activated sludge samples taken from processes operating at sludge ages varying from 2,5 to 30 days and at temperatures varying from 14°C to 20°C. Their findings indicated that the mass fraction of TKN in organic sludge was apparently independent of the operational conditions of the activated sludge process and remained approximately

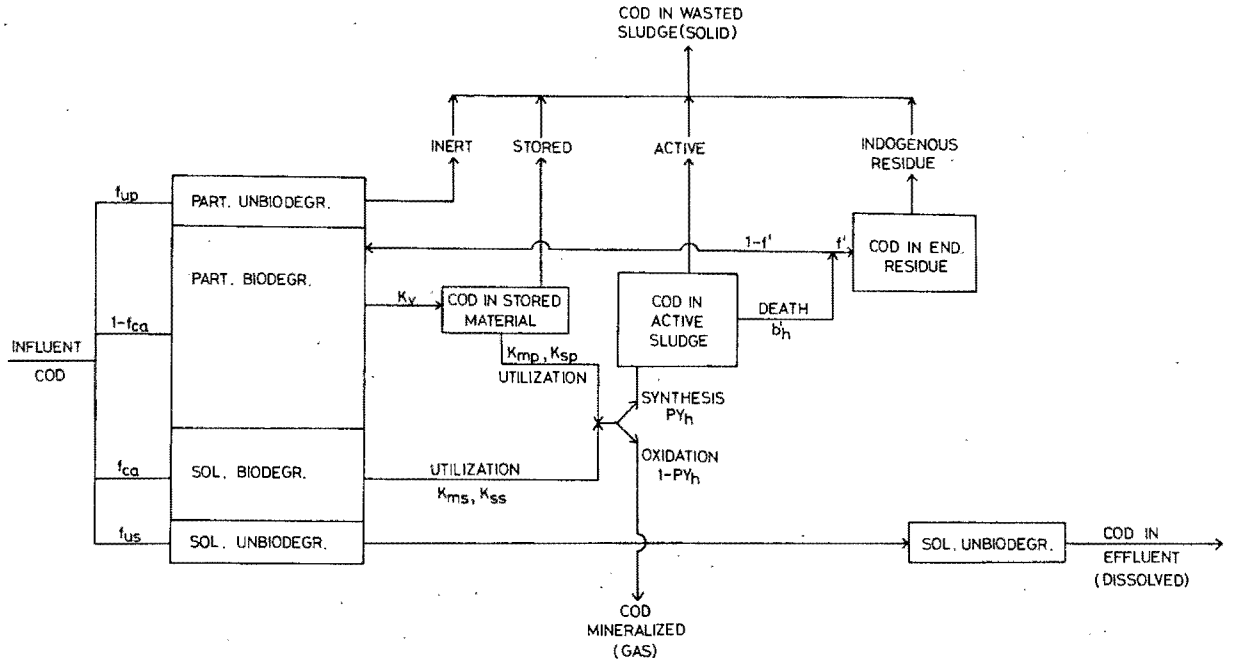

Fig. 3.2 *Schematic representation of the mechanisms associated with utilization of organic material and heterotrophic sludge growth and death in an aerobic activated sludge process.*

constant at $f_n = 0,1$ mg TKN.mg VSS^{-1}. The constancy of f_n, despite the fact that the proportions of active material, endogenous residue and inert material change significantly with sludge age led them to assume that the mass fractions of TKN in the different fractions of organic sludge was the same, i.e., $f_n = 0,1$ mg TKN.mg VSS^{-1} for X_a, X_e, and X_i. This does not appear to be unreasonable in view of the fact that based on this assumption, when balances are performed over an activated sludge process, consistent experimental nitrogen recoveries of 95 to 100 per cent are obtained.

5. UTILIZATION OF NITROGENOUS MATERIAL AND AUTOTROPHIC SLUDGE GROWTH

5.1 Characterization of nitrogenous material in wastewater

Similar to the division of organic material in wastewater into fractions - (Section 3.3.1), it is also possible to divide the TKN in wastewater into different fractions. In the model developed by Dold *et al* four influent TKN fractions are distinguished:

(1) Unbiodegradable soluble nitrogen, N_{ui}

(2) Unbiodegradable particulate nitrogen, N_{pi}

(3) Ammoniacal nitrogen, N_{ai}

(4) Organic nitrogen (biodegradable), N_{oi}

5.1.1. Unbiodegradable soluble nitrogen fraction

Similar to the unbiodegradable soluble COD fraction this fraction of the TKN is not affected by the activity of either heterotrophic or autotrophic bacteria and is discharged unchanged with the effluent. Dold *et al* (1980) found that in domestic sewage this fraction of the influent TKN, N_{ti}, is so small that usually it may be neglected.

5.1.2. Unbiodegradable particulate nitrogen fraction

From the conclusions set out in Section 4.2 Marais and Ekama (1976)

accepted that a fraction of the influent TKN, N_{ti}, is associated with unbiodegradable particulate organic influent material, X_{ii}:

$$N_{pi} = f_n X_{ii} = f_n f_{up} S_{ti}$$

where

N_{pi} = unbiodegradable particulate influent TKN concentration

f_n = mass fraction of nitrogen in organic solids

= 0,1 mg N.mg VSS.$^{-1}$

Usually with normal values of f_{up} = 0,04 (settled sewage) and 0,09mg VSS.mg COD^{-1} (raw sewage) the concentration of N_{pi} is about 5 to 10 per cent of the total influent TKN, N_{ti}.

5.1.3 Ammoniacal nitrogen fraction

In domestic sewage about 65 to 85 per cent of the total TKN is present in the form of ammonia. The magnitude of the fraction appears to depend on the sewer system retention time and temperature: Long sewer lines with flat gradients can retain the sewage for considerable periods of time and allow hydrolysis of organic nitrogen to ammonia (Ammonification); this reaction will be enhanced at higher temperatures.

5.1.4 Organic nitrogen fraction (biodegradable)

This fraction of the influent TKN is defined to be composed of nitrogen in organic substances such as urea and aminoacids, excluding however, the nitrogen associated with unbiodegradable particulate organic material, N_{pi}. Usually the concentration of organic biodegradable TKN in the influent N_{oi} is about 15 per cent of the total influent TKN, N_{ti}, but due to biological action in the sewer lines, this fraction may vary depending on the retention time and temperature

in the sewer lines. Different from ammoniacal nitrogen (which is a well defined monosubstrate) organic biodegradable material comprises a large number of compounds ranging from small soluble molecules such as urea and aminoacids to large macromolecules such as proteins. Some of these can pass directly through bacterial cell walls, whereas others need to be adsorbed and broken down by extracellular enzyme in order to pass. At present, however, it is not possible to distinguish the different fractions of organic biodegradable material and the pathways of their use and degradation. Consequently, for the purpose of modelling organic biodegradable nitrogen, Dold *et al* (1980) approximated an equivalent uniform compound that is involved in several mechanisms that take place in the activated sludge process (see Section 5.6).

Summarizing from the preceding discussion the following formulations for the division of the fractions of influent nitrogenous material in wastewater, N_{ti}, are made (Dold *et al* 1980).

$$N_{ui} = f_{un}.N_{ti} \quad (3.43)$$

$$N_{pi} = f_{n}.f_{up}.S_{ti} \quad (3.44)$$

$$N_{ai} = f_{na}.N_{ti} \quad (3.45)$$

$$N_{oi} = (1-f_{na}-f_{un})\ N_{ti} - f_{n}.f_{up}.S_{ti} \quad (3.46)$$

where

N_{ti} = total influent TKN (mg N.ℓ^{-1})

N_{ui} = unbiodegradable soluble TKN (mg N.ℓ^{-1})

N_{pi} = unbiodegradable particulate TKN (mg N.ℓ^{-1})

N_{ai} = ammoniacal nitrogen (mg N.ℓ^{-1})

N_{oi} = organic nitrogen (biodegradable) (mg N.ℓ^{-1})

f_{un} = fraction (unbiodegradable soluble : total TKN)

= 0,00 to 0,02 mg N.mg N^{-1}

f_n = mass ratio (nitrogen : volatile solids)

= 0,1 mg N.mg VSS^{-1}

f_{na} = fraction (ammoniacal : total TKN)

= 0,65 to 0,85 mg N.mg N^{-1}

5.2 Composition of autotrophic sludge

Autotrophic sludge in the activated sludge process comprises the biomasses of two genera of of autotrophic bacteria, namely Nitrosomonas and Nitrobacter, that mediate the oxidation of ammonia to nitrite and nitrite to nitrate respectively. The growth of both genera is proportional to the masses of oxidized substrates, i.e:

$$dX_n = Y_n . d N_a \quad (3.47)$$

$$dX_{ni} = Y_{ni} . d N_{NO_2} \quad (3.48)$$

where

X_n = concentration of Nitrosomonas (mg VSS.ℓ^{-1})

Y_n = yield coefficient for Nitrosomonas the oxidation of ammonia to nitrite (mg X_n.mg NH_3-$N.^{-1}d^{-1}$)

N_a = concentration of ammonia (mg N.ℓ^{-1})

N_{NO_2} = concentration of nitrite (mg N.ℓ^{-1})

X_{ni} = concentration of Nitrobacter (mg VSS.ℓ^{-1})

Y_{ni} = yield coefficient of the Nitrobacter upon oxidation of nitrate to nitrite.

The growth characteristics of these autotrophic bacteria have been discussed in Chapter 2. From Table 2.1 it may be noted that the yield coefficient of Nitrosomonas is much larger than that of nitrobacter. This means that the mass of autotrophic sludge is

almost exclusively composed of Nitrosomonas. Later it will be shown that the mass of Nitrosomonas itself is insignificant compared to the mass of heterotrophic sludge in active sludge processes treating domestic wastes. For these reasons the concentrations of nitrifiers (Nitrosomonas and Nitrobacter) are not included in the expression of the volatile solids concentration X_v (Chapter 2, Section 3.5.2). It will also be considered that the mass of nitrogen required for growth of autotrophic sludge is so small that it may be neglected (Dold *et al* 1980).

5.3 Utilization of ammonia

Ammonia in the activated sludge process is used by both heterotrophic and autotrophic microorganisms.

5.3.1 Utilization of ammonia for heterotrophic sludge synthesis

In the activated sludge process nitrogen is removed from the liquid phase and incorporated in active sludge where it is converted into proteinaceous material. Although it is possible for the active organisms to utilize amino-acids (containing organic nitrogen) directly for synthesis, to quantify this behaviour would be very difficult. Consequently the more usual course is accepted that the organisms utilize ammonia only in the synthesis reaction. With a mass fraction, f_n, in active sludge, the rate of utilization of ammonia for heterotrophic sludge synthesis is proportional to the rate of heterotrophic sludge synthesis:

$$r_{nh} = -(dN_a/dt)_{nh} = f_n.r_g = f_n Y_h (r_{us}+r_{up}) \qquad (3.49)$$

where

r_{nh} = rate of utilization of ammonia for heterotrophic sludge synthesis (mg $N.\ell^{-1}.d^{-1}$)

r_g = rate of active sludge synthesis (mg $VSS.\ell^{-1}.d^{-1}$)

f_n = mass fraction of nitrogen in organic sludge.

$= 0{,}1 \text{ mg N.mg VSS}^{-1}$

Index "nh" refers to nitrogen requirement for heterotrophic sludge synthesis.

5.3.2 Utilization of ammonia for nitrification

In the discussion of nitrification (chapter 2) it was established that the kinetics of this process could be described by the Monod equation. Considering the oxidation step from ammonia to nitrite by Nitrosomonas as rate limiting and the oxidation of nitrite to nitrate as immediate (i.e., considering the nitrification process as equivalent to the oxidation of ammonia to nitrate) one has:

$$r_n = -(dN_a/dt)_n = (dN_n/dt)_n \qquad (3.50)$$

$$= K_{nsT}.N_a/(N_a+K_{nT}).X_n$$

where

r_n = rate of oxidation of ammonia

= nitrification rate

= rate of production of nitrate (mg $N.\ell^{-1}d^{-1}$)

N_a = ammonia concentration (mg $N.\ell^{-1}$)

N_n = nitrate concentration (mg $N.\ell^{-1}$)

K_{nsT} = specific utilization rate constant of ammonia by Nitrosomonas (mg NH_3-$N.\ell^{-1}d^{-1}$)

K_{nT} = half saturation value for nitrification (mg NH_3-$N.\ell^{-1}$)

X_n = concentration of Nitrosomonas (mg $VSS.\ell^{-1}$)

index "n" refers to nitrification.

Note that in Eq. (3.50) the rate of decrease of the ammonia concentration

for nitrification is equated to the rate of increase of the nitrate concentration. This implies that the approximation is made that all ammonia nitrogen in the nitrification is used for oxidation and that the mass required for synthesis of Nitrosomonas and Nitrobacter is neglected. The error in doing so is very small (about 1 per cent) as discussed in Chapter 2.

5.4 Oxygen requirement for Nitrification

In Chapter 2 (section 2.2.1) from stoichiometric considerations it was calculated that the oxygen requirement for oxidation of ammonia nitrate is 4,57 mg O_2/mg N. i.e.,

$$r_{on} = (dO/dt)_n = 4{,}57 . r_n \qquad (3.51)$$

where

r_{on} = oxygen uptake rate for nitrificaion (mg $O.\ell^{-1}d^{-1}$)

O = dissolved oxygen concentration

Index "n" refers to nitrification.

5.5 Growth and death of Nitrifiers

In Chapter 2 it was shown that only the growth behavior of Nitrosomonas is of practical importance in the activated sludge process and that the growth of Nitrosomonas is proportional to the mass of ammonia oxidized: the proportionality constant being the yield coefficient for Nitrosomonas Y_n. Hence:

$$r_{gn} = Y_n r_n = Y_n K_{nst} . N_a/(N_a+K_{nt}) \quad X_n \qquad (3.52)$$

where

r_{gn} = rate of growth of nitrifiers.

Y_n = yield coefficient of nitrifiers.

The formulation in Eq (3.52) follows the practice widely used in sanitary engineering, where the sludge growth is expressed in terms of yield coefficient (Y_n) and a substrate utilization rate constant (K_{nsT}). Bacteriologists tend to use the maximum specific growth rate constant μ_{nmT}, i.e. $\mu_{nmT} = Y_n.K_{nsT}$ or $K_{nsT} = \mu_{nmT}/Y_n$. In nitrification kinetics the custom appears to be to utilize μ_{nmT} instead of $Y_n K_{nsT}$ and all the work in this exposition will be presented accordingly, i.e., the growth rate of nitrifiers (Eq (3.52)) is rewritten as:

$$r_{gn} = \mu_{nmT}.N_a/(N_a+K_{nT}) \quad X_n \tag{3.54}$$

where

μ_{nmT} = maximum specific growth rate for nitrifiers (mg X_n. mg $X_n^{-1}.d^{-1}$)

Dold *et al* (1980) accepted that Nitrosomonas die off at a rate proportional to their concentration, i.e.,

$$r_{dn} = b_{nT}.X_n \tag{3.55}$$

where

r_{dn} = death rate of nitrifiers mg$X_n.\ell^{-1}.d^{-1}$

b_{nT} = death rate constant for nitrifiers (mg X_n.mg $X_n^{-1}.d^{-1}$)

= 0,04 $(1,029)^{T-20}$ mg X_n.mg $X_n^{-1}.d^{-1}$

Acceptance of a death rate for nitrifiers is reasonable as it is a phenomenon present in all biological life. The magnitude of the death rate constant is uncertain as it can only be determined in pure cultures of Nitrosomonas and the constant thus obtained does not necessarily apply to mixed cultures such as in activated sludge processes. However, in Chapter 2 it was pointed out that the numerical value of the death rate constant for nitrifiers is not of

great importance for the kinetics of nitrification.

5.6 Reactions of organic biodegradable nitrogen

Modelling of the reactions involving organic biodegradable nitrogen in the activated sludge process has been difficult for three reasons: (1) The concentration of organic nitrogen in the activated sludge process is usually quite small and the determination of its concentration is relatively inaccurate by the very nature of the required analytical procedures; (2) Organic biodegradable nitrogen comprises a wide spectrum of chemical compounds that are not likely to react uniformly in the activated sludge process; (3) Several mechanisms affecting the organic biodegradable nitrogen occur simultaneously and are interdependent so that it is difficult to establish the kinetics of each mechanism individually. Dold *et al* distinguished four mechanisms that influence the concentration of organic nitrogen in the activated sludge process:

(a) Lysis of organic nitrogen from dead heterotrophs.

(b) Conversion of organic nitrogen to ammonia.

(c) Adsorption of organic nitrogen, accompanying the storage of particulate biodegradable organic material.

(d) Desorption of organic nitrogen due to utilization of stored organic material.

(a) Lysis of organic nitrogen from dead hetertrophs:

In the death regeneration approach organic nitrogen is generated internally in the activated sludge process due to death and lysis of heterotrophic organisms. Considering that all the nitrogen in sludge is organic nitrogen and that a fraction (1-f') of the dead organisms is lysed, having a nitrogen mass fraction f_n, to the liquid phase, the rate of internal generation of organic nitrogen is:

$$r_{nl} = (d\,N_o/dt)_l = f_n(1-f')r_d \qquad (3.56)$$

r_{nl} = rate of increase of the organic biodegradable nitrogen concentration due to lysis. (mg $N.\ell^{-1}d^{-1}$)

Index "1" refers to lysis.

(b) Conversion of organic nitrogen to ammonia.

From comparison of the influent and the effluent concentrations of organic nitrogen in steady state activated sludge processes Marais and Ekama (1976) concluded that in the activated sludge process organic nitrogen is converted to ammonia. They suggested the following model to describe the rate of conversion:

$$r_{co} = (dN_o/dt)_{co} = K_{rT}.N_o.X_a \qquad (3.57)$$

where

r_{co} = rate of conversion of organic nitrogen to ammonia (mg $N.\ell^{-1}.d^{-1}$)

N_o = concentration of organic nitrogen (mg $N.\ell^{-1}$)

X_a = concentration of active sludge (mg $VSS.\ell^{-1}$)

K_{rT} = conversion rate constant (ℓ.mg $VSS^{-1}.d^{-1}$)

Index "co" refers to conversion of organic biodegradable nitrogen to ammoniacal nitrogen.

(c) and (d) Adsorption and desorption.

Ekama and Marais (1978) found that organic nitrogen was removed from the liquid phase when storage of particulate slowly biodegradable material occurred. Dold *et al* formulated the following model for changes of organic nitrogen concentration due to storage and utilization of particulate slowly biodegradable material: (1) Per unit of particulate biodegradable COD there is a proportional amount of organic nitrogen that is also stored on the active sludge. When utilization of the stored material occurs the associated stored organic nitrogen is released to the liquid phase. In terms of this

behaviour organic nitrogen is not taken directly by the cell mass; ammonia is taken up for synthesis as indicated in Eq (3.49). Accepting a proportionality constant f_{ns} for the ratio of (stored organic nitrogen : stored slowly biodegradable organic material) one has for the process of adsoprtion or storage:

$$r_{ns} = -(dN_o/dt)_s = f_{ns}.r_a \qquad (3.58)$$

where

r_{ns} = rate of decrease of concentration of organic nitrogen due to storage (mg $N.\ell^{-1}.d^{-1}$)

Index "s" refers to storage

When utilization of the stored material occurs organic nitrogen is released back to the liquid phase:

$$r_{nd} = (dN_o/dt)_d = f_{ns}.r_{up} \qquad (3.59)$$

r_{nd} = rate of increase of concentration of organic nitrogen due to desorption of stored organic nitrogen upon utilization of slowly biodegradable organic material.

Index "d" refers to desorption.

While it is unlikely that this model proposed by Dold *et al* is a true description of the physical reality of storage and synthesis of nitrogenous material, from the close correspondence between predicted results using the model and experimental results under cyclic flow and load conditions reported by Dold *et al* and from experiments later in the present work (Chapter 4), it is apparent that the model gives an adequate mathematical description and is therefore accepted.

Dold *et al* (1980) suggested a numerical value of f_{ns} as the ratio (influent organic biodegradable TKN)/influent slowly biodegradable organic material) i.e., $f_{ns} = N_{oi}/S_{bpi}$. The experiments presented in Chapter 4 however would indicate that this value is too low. In this respect it must be remembered that in the formulation of Dold *et al* only a fraction of organic biodegradable nitrogen enters the

activated process with the influent; the rest being generated internally as a result of lysis of dead organisns. For this reason, a more appropriate value for f_{ns} would appear to be the weighted average of the ratios (influent organic biodegradable TKN)/ influent slowly biodegradable material) and of (organic biodegradable nitrogen in lysed cells /slowly biodegradable material in lysed cells) i.e.,

$$f_{ns} = \frac{f_{ns1}.rate_1 + f_{ns2}.rate\ 2}{rate\ 1 + rate\ 2} \qquad (3.59a)$$

where

f_{ns1} = influent organic biodegradable material /influent slowly biodegradable material

= N_{oi}/S_{bpi}

f_{ns2} = organic biodegradable material from lysed cells /slowly biodegradable material from lysed cells

= $f_n(1-f')/P(1-f') = f_n/P$

$rate_1$ = rate of introduction of influent organic nitrogen

= $Q.N_{oi}/V$ mg $N.\ell^{-1}h^{-1}$

$rate_2$ = rate of internal generation of organic nitrogen

= $f_n(1-f)r_d$

Simulations based on this definition of the proportionality constant f_{ns} gave good correspondence between theoretical predictions and the experimental work reported by Dold *et al* as well as that reported in Chapter 4 of the present work and was therefore accepted.

5.7 Basic equations for reactions of nitrogenous material and for autotrophic sludge growth

The differential equations that describe the reaction rates of

the different nitrogenous species in the aerobic activated sludge process can now be expressed in terms of the basic mechanisms described in Sections 5.3 to 5.6. These mechanisms are repeated below.

The following mechanisms are distinguished:

(1) Utilization of ammonia by heterotrophs:

$$r_{nh} = f_n Y_h (r_{us}+r_{up})$$

$$= [f_n Y_h K_{msT}.S_{bs}/(S_{bs}+K_{ssT}) + K_{mpT} PX_S+K_{spT} X_a)] X_a \quad (3.49)$$

(2) Utilization of ammonia for nitrification:

$$r_n = [K_{nsT}.N_a/(N_a+K_{nT})] X_n$$

$$= [(\mu_{nmT}/Y_n) N_a/(N_a+K_{nT})] X_n \quad (3.50)$$

(3) Synthesis of nitrifiers:

$$r_{gn} = [\mu_{nmT} N_a/(N_a+K_{nT})] X_n \quad (3.53)$$

(4) Death of nitrifiers:

$$r_{dn} = [b_{nT}] X_n \quad (3.55)$$

(5) Conversion of organic nitrogen:

$$r_{co} = [K_{rT}N_o] X_a \quad (3.57)$$

(6) Lysis of organic nitrogen:

$$r_{nl} = f_n(1-f') r_d \quad (3.56)$$

$$= [f_n(1-f') b'_{hT}] X_a$$

(7) Storage of organic nitrogen:

$$r_{ns} = f_{ns} r_a$$

$$= [f_{ns}.K_{vt}\ S_{bp}\ (F_{ma} - X_s/X_a)]\ X_a \qquad (3.58)$$

(8) Desorption of organic nitrogen:

$$r_{nd} = f_{ns} r_{up}$$

$$= [f_{ns}\ K_{mp}\ PX_s/(PX_s + K_{spT}\ X_a)]\ X_a \qquad (3.59)$$

With the aid of these basic mechanisms the reaction rates of the different nitrogenous components and of nitrifying sludge can now be expressed as follows:

Ammoniacal nitrogen:

$$r_{na} = (dNa/dt)_r = r_{co} - r_{nh} - r_n$$

$$= [K_{rT}.N_o - f_n Y_h (K_{msT}.S_{bs}/(S_{bs}+K_{ssT}) + K_{mpT}\ X_s P/(X_s P + K_{spT} X_a)]\ X_a$$

$$- [(\mu_{nmT}/Y_n).N_a/(N_a+K_{nT})]\ X_n \qquad (3.60)$$

Nitrate nitrogen:

$$r_{nn} = (d\ N_n/dt)_r = r_n$$

$$= [(\mu_{nmT}/Y_n)\ N_a/(N_a+K_{nT})]\ X_n \qquad (3.61)$$

Organic nitrogen:

$$r_{no} = (dN_o/dt)_r = - r_{co} - r_{ns} + r_{nd} + r_{nl}$$

$$= [- K_{rT}.N_o + f_{ns}(K_{vT}.S_{bp}(F_{ma} - X_s/X_a) - K_{mpT} X_s P/(X_s P + K_{spT} X_a))$$

$$+ f_n (1-f') b'_{hT}]\ X_a \qquad (3.62)$$

Nitrosomonas:

$$r_{xn} = (dX_n/dt)_r = r_{gn} - r_{dn}$$

$$= [\mu_{nmT}\ N_a/(N_a+K_{nT}) - b_{nT}]\ X_n \qquad (3.63)$$

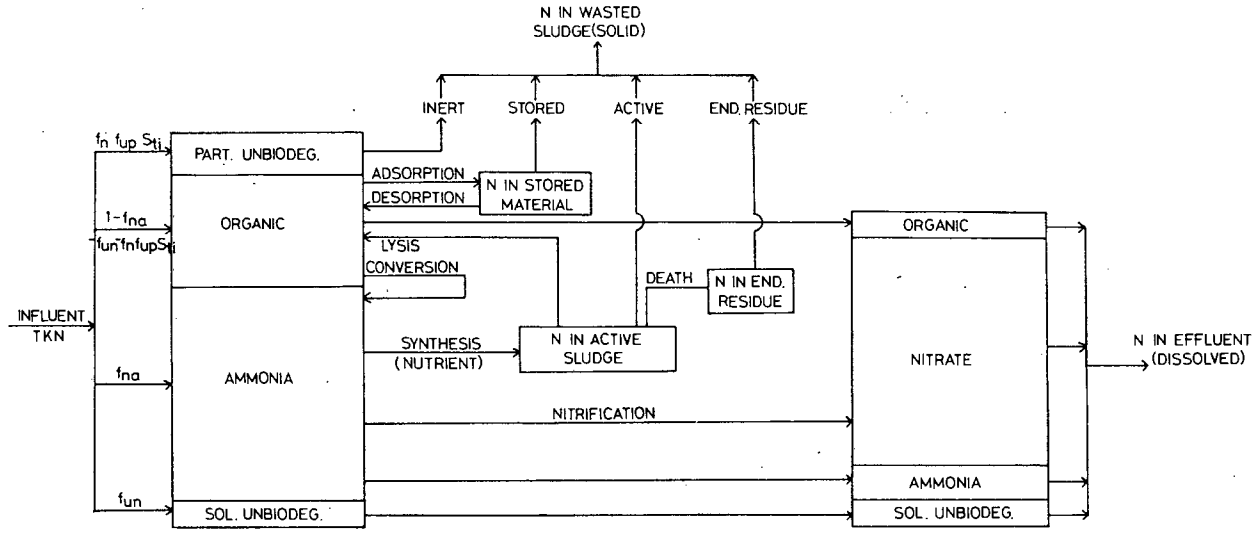

Fig. 3.3 *Schematic representation of the mechanisms associated with the nitrogenous material in the aerobic activated sludge process.*

Oxygen consumption for nitrification:

$$O_n = (dO/dt)_n = 4{,}57\ r_n$$

$$= [4{,}57\ (\mu_{nmT}/Y_n)\ N_a/(N_a+K_{nT})]\ X_n \qquad (3.64)$$

Fig. 3.3 shows a schematic representation of the different reactions of nitrogenous material in the aerobic activated sludge process and of the different forms under which nitrogen leaves the process aerobic activated sludge process.

6. ALKALINITY CHANGES IN THE ACTIVATED SLUDGE PROCESS

In Chapter 2, Section 3.1 from stoichiometric considerations it was shown that oxidation of organic material under aerobic conditions does not affect the alkalinity of mixed liquor; oxidation of nitrogenous material, however, does lead to changes in alkalinity. In aerobic processes the changes of alkalinity are due to two simultaneous reactions: (1) nitrification, and (2) ammonification of organic nitrogen.

(1) Nitrification

It was shown in Chapter 2 that stoichiometrically there is a decrease of 7,14 mg alkalinity (as Ca CO_3) when 1 mg NH_4 - N is nitrified to NO_3 - N i.e.,

$$r_{aln} = (d\ Alk/dt)_n = -7{,}14\ r_n \qquad (3.65)$$

where

r_{aln} = rate of change of alkalinity due to nitrification (mg $CaCO_3.\ell^{-1}.d^{-1}$)

Index "n" refers to nitrification.

(2) Ammonification

The rate of ammonification is the nett rate of conversions of organic nitrogen to ammonia: In the formulations of the previous sections (Sections 5.6 and 5.3.1) two rates were identified, a rate r_{co} of conversion of organic nitrogen to ammonia (Eq (3.56)) and simultaneously a rate r_{nh} of ammonia conversion to organic nitrogen during synthesis of heterotrophic organisms (Eq (3.49)).
Hence the nett rate of conversion of organic nitrogen to ammonia is expressed as:

$$r_{am} = r_{co} + r_{nh} \tag{3.66}$$

where

r_{am} = rate of ammonification.

In Chapter 2 it has been shown that stoichiometrically upon the conversion of 1 mg N of organic nitrogen to 1 mg. NH_4^+- N there is an increase in alkalinity of 3,57 mg $CaCO_3$. Hence:

$$r_{ala} = (dAlk/dt)_{am} = 3{,}57\, r_{am} = 3{,}57\, (r_{co} - r_{nh}) \tag{3.67}$$

where

r_{ala} = rate of change of alkalinity due to ammonification of organic nitrogen (mg $CaCO_3.\ell^{-1}.d^{-1}$)

Index "am" refers to ammonification.

The net rate of change of alkalinity is the sum of the rates of the two reactions:

$$r_{alk} = r_{aln} + r_{ala} \tag{3.68}$$

$$\begin{aligned} r_{alk} &= -7{,}14\, r_n + 3{,}57\, (r_{co} - r_{nh}) \\ &= [-7{,}14(\mu_{nmT}/Y_n)Na/(Na+K_{nT})]\, X_n \\ &\quad +3{,}57\, [K_{rt}N_o - f_n Y_h (K_{msT}S_{bs}/(S_{bs}+K_{ssT}) + K_{mpT}X_s P/(X_s P + K_{spT}X_a))]\, X_a \end{aligned} \tag{3.69}$$

where

r_{alk} = rate of change of alkalinity due to nitrification and ammonification (mg $CaCO_3 . \ell^{-1} . d^{-1}$)

Equation (3.69) describes the rate of change of alkalinity in the aerobic activated sludge process provided two assumptions are made:

1. No precipitation or dissolution of $CaCO_3$ or other compounds that affect the alkalinity take place.
2. The abstraction of phosphates from the liquid phase into the solid phase for sludge growth, and the hydrolysis of organic phosphates to orthophosphates do not affect the alkalinity.

With regard to the first assumption: This is normally justified in the activated sludge process because (a) influent sewage usually does not contain calcium carbonate or other solids that upon dissolution affect the alkalinity, and (b) in the activated sludge process usually there is a decrease of alkalinity so that it is unlikely that precipitation of $CaCO_3$ will occur.

With regard to the effects of phosphorous transformations, when P acts as a nutrient in sludge growth, incoporation of PO_4 in the active mass will decrease the alkalinity whereas hydrolysis of organic phosphate will increase the alkalinity. The alkalinity changes however, usually, are very small when compared with the changes due to nitrificaiton or ammonification.

In the next Chapter experimental data will be presented, indicating that changes of mixed liquor alkalinity in an aerobic environment can be predicted very closely if only changes due to nitrification and ammonification are considered.

7. pH CHANGES IN THE ACTIVATED SLUDGE PROCESS

In this section it is the intention to show theoretically that in the activated sludge process a relatively stable pH will be maintained provided the alkalinity in the mixed liquor is maintained above 35 ppm $CaCO_3$; for alkalinities below this a rather unstable mixed liquor pH may be expected. The effect of alkalinity changes on the pH of mixed liquor can be analysed in terms of the model developed by Loewenthal and Marais (1976). This model describes the interrelationships between alkalinity, acidity and pH in aquous solutions containing weak acid-base systems. In order to simplify the discussion two assumptions will be made: (1) There are no changes in alkalinity due to precipitation or dissolution of salts, and (2) The alkalinity and acidity are due only to the $CO_2 - HCO_3^- - CO_3^=$ equilibria in the mixed liquor, i.e., the contributions of the equilibria $NH_3 - NH_4^+$ and $H_3PO_4 - H_2PO_4^- - HPO_4^= - PO_4^{\equiv}$ to the total alkalinity are neglected. This is justified in the activated sludge process because (a) the pH of the mixed liquor is normally much lower than the pK value of the equilibrium $NH_3 - NH_4^+$ ($pK_{NH_3} = 9,3$) so that ammoniacal nitrogen is present almost exclusively in the form of NH_4^+ i.e. it behaves as a monovalent cation and does not affect the alkalinity, and (b) the concentration of phosphates is so low that it does not normally affect the mixed liquor alkalinity in any significant manner. The carbonate system is completely defined by any two of a number of system parameters. The most useful ones are alkalinity and pH because these can be determined accurately by simple analytical procedures. The definitions of alkalinity and pH are respectively:

$$Alk = [HCO_3^-] + 2\,[CO_3^=] + [OH^-] - [H^+]$$

where

$[X]$ = concentration of X in moles.ℓ^{-1}

$$pH = -\log a_H = -\log(F_m [H^+])$$

where

a_H = activity of the hydrogen ion.

F_m = activity coefficient of the hydrogen ion.

The relationship between the concentrations of the different inorganic carbon species and the pH can be established from the dissociation reactions of CO_2 in water:

$$H_2CO_3^* \overset{k_1}{\rightleftarrows} HCO_3^- + H^+ \qquad (3.73)$$

$$HCO_3^- \overset{k_2}{\rightleftarrows} CO_3^- + H^+ \qquad (3.74)$$

From Eq (3.73):

$$(HCO_3^-) = (H_2CO_3^*)\ k_1/(H^+)$$

$$[HCO_3^-] = [H_2CO_3^*]\ \frac{k_1}{F_m}\left[\frac{1}{H^+}\right] = [H_2CO_3^*]\ \frac{k_1'}{[H^+]} \qquad (3.75)$$

where

(X) = activity of X (moles.ℓ^{-1})

$[X]$ = concentration of X (moles.ℓ^{-1})

F_m = activity coefficient for monovalent ions in the mixed liquor, i.e. hydrogen and bicarbonate ions.

k_1 = thermodynamic dissociation constant

= $4{,}45.10^{-7}$ (20^oC)

k_1' = actual dissociation constant taking into account the activities of the ions.

Similarly from Eq (3.72)

$$(CO_3^{=}) = (HCO_3^{-})\ k_2/(H^+)$$

$$[CO_3^{=}] = [HCO_3^{-}]\ \frac{k_2}{F_d}\left[\frac{1}{H^+}\right] = [H_2CO_3^*]\left[\frac{k_1'}{H^+}\right]\left[\frac{k_2'}{H^+}\right] \qquad (3.76)$$

where

F_d = activity coefficient for a divalent ion in the mixed liquor (i.e. the carbonate ion)

k_2 = thermodynamic dissociation constant.

= $4{,}69.10^{-11}$ (20^oC)

k_2' = actual dissociation constant.

Equations (3.74 and 3.75) can now be substituted in the expressions for alkalinity and acidity (Eq (3.70) and (3.71) respectively):

$$Alk = [H_2CO_3^*]\left[\frac{k_1'}{H^+}\right] + \left[\frac{2k_1'k_2'}{H^{+2}}\right] + \left[\frac{k_w'}{H^+}\right] - [H^+] \qquad (3.77)$$

In the activated sludge process it is to be expected that the concentration of $(H_2CO_3)^*$ (i.e., of dissolved carbon dioxide) has some constant value depending upon the rate of production of CO_2 from oxidized organic material and the stripping efficiency of the aeration system. If it is assumed that the concentration of $(H_2CO_3)^*$ (i.e., of dissolved carbon dioxide) is constant, changes of the hydrogen ion concentration (and hence of pH) due to changes in alkalinity can be calculated from Eq (3.77). The relationship between pH and alkalinity is shown plotted in Fig. 3.4 for a series of fixed CO_2 concentrations ranging from 0,5 mg CO_2/ℓ (i.e. the saturation value) to 10 mg CO_2/ℓ (i.e., 20 times supersaturated). A temperature of 20^oC and concentration of inorganic solids of 1000 $mg.\ell^{-1}$ (i.e., an ionic strength of 0,01 and activity coefficients F_m = 0,90 and F_d = 0,67 for mono and divalent ions respectively) have been assumed.

From Fig. 3.4 it can be noted that for a particular concentration of CO_3 change of pH with alkalinity is not very significant for alkalinities greater than about 35 ppm $CaCO_3$; an increase from 35 to 400 ppm $CaCO_3$ results in an increase in pH of less than 1 unit.

In contrast for alkalinities less than 35 ppm $CaCO_3$ there is a strong dependency of the pH on alkalinity; the pH may readily decrease to a value smaller than 7 which, as shown in Chapter 2, will cause a rapid decline in the nitrification rate and may lead to a complete collapse of nitrification. Hence for efficient and reliable

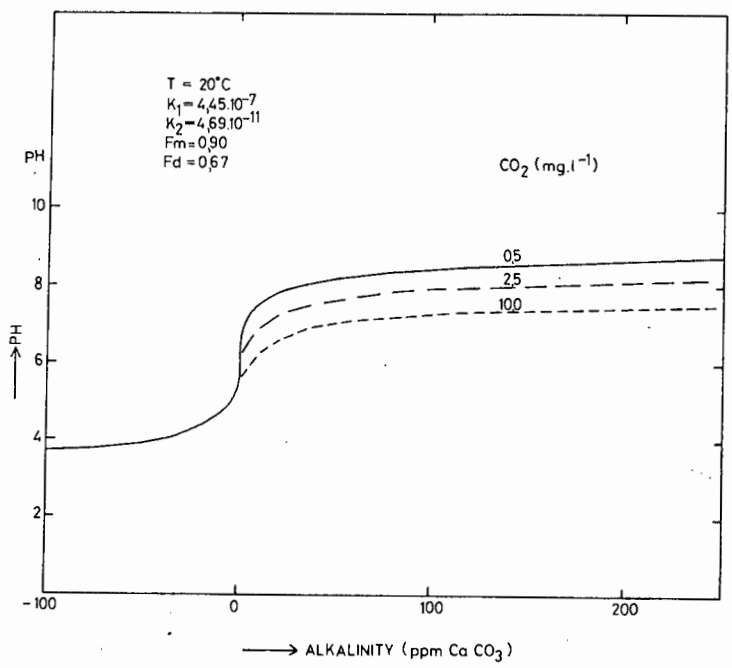

Fig. 3.4 *Mixed liquor pH versus mixed liquor alkalinity for different dissolved CO_2 concentrations.*

nitrification it is necessary that the alkalinity of the mixed liquor is not less than about 35 ppm. In a study of nitrification behaviour Haug and McCarty (1970), from experimental observations, arrived at the same minimum value of alkalinity as derived above theoretically. Implications on process design will be discussed in Section 9.

8. PROCESS EQUATIONS FOR THE ACTIVATED SLUDGE PROCESS

In Sections 3.1 to 3.7 of this Chapter, equations for the reaction rates of utilization of biodegradable organic substrate and active sludge growth and death as described in the model developed by Dold *et al* were discussed. Equations for the reaction rates of nitrogenous material and for growth and death of nitrifiers described in the same model were discussed in Sections 5.1 to 5.7 of this Chapter.

In Sections 6 and 7 it has been shown that, using the equations developed by Dold *et al* equations for reaction rates for alkalinity and for mixed liquor pH can be developed from stoichiometric considerations.

The kinetic model by Dold *et al* extended to include alkalinity, is characterized by a set of simultaneous non linear differential equations. These equations are brought together in Table 3.1. Equations in Table 3.1 refer only to the reaction rates of the variables S_{bs}, S_{bp}, X_s, X_a, X_n, N_a, N_o, N_n, and Alk.

Table 3.1 Basic differential equations for the aerobic activated sludge process.

Let

$$r_{us} = [K_{msT} \cdot S_{bs}/(S_{bs}+K_{ssT})]\ X_a \quad (3.78)$$

$$r_a = [K_{vT} \cdot S_{bp}(F_{ma}-X_s/X_a)]\ X_a \quad (3.79)$$

$$r_{up} = [K_{mpT} \cdot X_s \cdot P)/(X_s P+K_{spT} X_a)]\ X_a \quad (3.80)$$

$$r_d = [b'_{hT}]\ X_a \quad (3.81)$$

$$r_n = [(\mu_{nmT}/Y_n)\ N_a/(N_a+K_{nT})]\ X_n \quad (3.87)$$

$$r_{dn} = [b_{nT}]\ X_n \quad (3.83)$$

$$r_{co} = [K_{rT}\ N_o]\ X_a \quad (3.84)$$

Then the reaction rates are:

$$r_{sbs} = -r_{us} \quad (3.86)$$

$$r_{sbp} = -r_a + P(1-f')\ r_d \quad (3.87)$$

$$r_{xs} = (r_s-r_{us})/P \quad (3.88)$$

$$r_{xa} = Y_h(r_{us}+r_{up}) - r_d \quad (3.89)$$

$$r_{xe} = f'r_d \quad (3.90)$$

$$r_{xn} = Y_n\ r_n - r_{dn} \quad (3.91)$$

$$r_{na} = -r_n = Y_h(r_{us}+r_{up}) + r_{co} \quad (3.92)$$

$$r_{nn} = r_n \quad (3.93)$$

$$r_{no} = -r_{co} - f_{ns}\ (r_a-r_{up}) + f_b(1-f')r_d \quad (3.94)$$

$$r_{alk} = -7{,}14\ r_n + 3{,}57\ (r_{co}-f_n\ Y_h(r_{us}+r_{up}) \quad (3.95)$$

$$O_c = (1-PY_h)\ (r_{us}+r_{up}) \quad (3{,}96)$$

$$O_n = 4{,}57\ r_n \quad (3.97)$$

$$O_t = O_c + O_n \quad (3.98)$$

$$r_{xv} = r_{sbp}/P + r_{xs} + r_{xa} + r_{xe} \quad (3.99)$$

In a particular activated sludge process the observed rates of change in the concentrations of the variables do not depend only on the kinetic expressions listed in Table 3.1, but also on the mixing regime and the hydraulic loading pattern of the reactor or reactors of the activated sludge process.

The mixing regime in any reactor can range between two extremes: completely mixed and plug flow regimes. In activated sludge processes the mixing regime in a particular reactor often can be described adequately by considering the reactor as equivalent to one or more completely mixed reactors in series. For this reason only the completely mixed regime will be considered. The rate of change in concentration of the variables identified in the previous sections of this Chapter in a completely mixed reactor is influenced by (1) the reaction rate, and (2) the hydraulic loading rate. A general expression for the rate of change of concentration for any of the variables in a particular reactor, j, of a series of completely mixed reactors can be written as:

$$(dC_j/dt) = r_{Cj} = r_{Cjr} + r_{Cjh} \qquad (3.100)$$

where

C_j = concentration of variable C in reactor j.

r_{Cj} = rate of change in the concentration of variable C in reactor j

r_{Cjr} = reaction rate of variable C in reactor j.

r_{Cjh} = rate of change in the concentration of variable C in reactor j due to the loading rate to and the abstraction rates from reactor j.

In a series reactor system the loading rate of a particular reactor may be due to an influent flow and recycle flows from any of the other reactors in the series and/or from the settler. The total flow entering the reactor is given by:

$$Q_{tot,j} = Q_j + Q_{i,j} + \sum_{n=j+1}^{NR} Q_{n,j} + Q_{s,j} \qquad (3.101)$$

$Q_{tot,j}$ = total flow entering reactor j.

Q_j = flow from reactor (j-1) and entering reactor j.

$Q_{i,j}$ = influent flow to reactor j.

$Q_{n,j}$ = mixed liquor recycle flow from reactor n to reactor j ($n > j$)

$Q_{s,j}$ = recycle flow from the settler to reactor j.

NR = number of reactors.

(all flows in $\ell.d^{-1}$)

The loading rate of variable C on the reactor is given by:

$$L_{Cj} = Q_j C_{j-1} + Q_{ij}.C_i + \sum_{n=j+1}^{NR} Q_{n,j} C_n + Q_{s,j}.C_s \qquad (3.102)$$

where

L_{Cj} = loading rate of variable C in reactor j ($mg.d^{-1}$)

C_i = influent concentration of C ($mg.\ell^{-1}$)

C_n = concentration of C in reactor n ($mg.\ell^{-1}$)

C_s = concentration of C in underflow recycle ($mg.\ell^{-1}$)

C_{j-1} = concentration of C in reactor (j-1) ($mg.\ell^{-1}$)

For a constant reactor volume V_j the total flow leaving the reactor Q_{je} must be equal to the total flow entering the reactor, $Q_{tot,j}$. Hence the mass flow from reactor j is given by the product $Q_{tot.}.C_j$. The nett rate of change of the mass of C in reactor j due to loading and abstraction can therefore be expressed as:

$$M_{Cjh} = V_j r_{Cjh} = Q_j.C_{j-1} + Q_{i,j} C_i + \sum_{n=j+1}^{NR} Q_{nj} C_n + Q_{s,j} C_s - Q_{tot.j}.C_j \qquad (3.103)$$

where

MC_{jh} = rate of accumulation of mass of variable C in reactor j due to influent and recycle flows.

r_{Cjh} = rate of change in the concentration of variable C in reactor j due to loading to and abstraction from that reactor.

Equation (3.103) can be used in Eq (3.100) to establish for each of the variables in each reactor, an expression for the rate of change in concentration of the variables in the reactor. However, it is not possible to write explicit expression for the concentrations as a function of time and space (reactor number) because the expressions for the reaction rates of Eqs (3.86 to 3.95) are non-linear differential equations that cannot be solved analytically. An analytical solution is possible only for the special case of the single reactor activated sludge system under constant flow and load conditions. In all other cases a solution can only be obtained by numerical integration of Eq (3.100). In Section 9, the analytical solution for the single reactor process will be discussed. Section 10 deals with the numerical integration procedures using a digital computer.

9. SINGLE REACTOR STEADY STATE AEROBIC ACTIVATED SLUDGE PROCESS

9.1 Solids retention time and liquid retention time

For a mathematical description of the single reactor completely mixed activated sludge process under constant flow and load conditions (Fig. 3.1) it is convenient to consider that (1) the mass of sludge retained in the settler is negligibly small compared to the mass in the reactor i.e., the settler is an instantaneous liquid-solid separator and (2) the sludge is wasted at constant rate with a flow q from the reactor.

With a volatile sludge concentration X_v the rate of wastage, r_w, is given as:

$$r_w = q.X_v \qquad (3.104)$$

where

r_w = rate of sludge wastage (mg VSS.d^{-1})

q = waste flow ($\ell.d^{-1}$)

Under steady state conditions the rate of sludge wastage is equal to the rate of sludge production. The sludge age, R_s, or the solids retention time can now be determined as:

$$R_s = \frac{\text{Mass of sludge present in the system}}{\text{Rate of sludge wastage}} \quad (d)$$

With a reactor volume V:

$$R_s = V.X_v/q.X_v = V/q \quad (d) \qquad (3.105)$$

Note that the solids retention time or sludge age is not related to the liquid retention time. The latter parameter, which determines the retention time of all soluble variables in the reactor is given by:

$$R_h = V/Q \qquad (3.106)$$

where

R_h = hydraulic retention time (d)

V = reactor volume (ℓ)

Q = influent flow. (ℓ/d)

9.2 Steady state balances

In the reactor under steady state conditions there is no nett change in the concentrations of any of the variables, i.e.

$$dC/dt = r_C = r_{Cr} + r_{Ch} = 0. \quad (3.107)$$

r_C = rate of change in concentration of the variables.

r_{Cr} = rate of change in concentration of the variables due to reaction.

r_{Ch} = rate of change in concentration of the variables due to hydraulic effects.

C refers to the concentrations of the independent variables discussed in the previous sections (S_{bs}, S_{bp}, X_s, X_a, X_e, X_n, N_a, N_n, N_o, and Alk) as well as the variables that are not effected by chemical reactions in the activated sludge process (S_u, X_i, N_u). For each of these variables a mass balance can be performed over the reactor (Fig 3.1):

$$V(dC/dt) = 0 = Q_i C_i - Q_e C_e - qC + r_{Cr}.V \quad (3.108)$$

where

(dC/dt) = rate of change of the variables (C) due to reaction and hydraulic effects. ($mg.\ell^{-1}d^{-1}$)

$Q_i C_i$ = mass loading rate of C with the influent flow ($mg.d^{-1}$)

$Q_e C_e$ = mass abstraction rate of C with the effluent flow ($mg.d^{-1}$)

qC = mass wastage rate with the waste flow ($mg.d^{-1}$)

Q_i = influent flow ($\ell.d^{-1}$)

Q_e = effluent flow ($\ell.d^{-1}$)

q = sludge waste flow ($\ell.d^{-1}$)

C = concentration of variable C in the reactor ($mg.\ell^{-1}$)

C_i = concentration of variable C in the influent ($mg.\ell^{-1}$)

C_e = concentration of variable in the effluent ($mg.\ell^{-1}$)

r_{Cr} = reaction rate of variable C in the reactor ($mg.\ell^{-1}d^{-1}$)

V = reactor volume (ℓ)

The difference between the influent flow Q_i and the effluent flow Q_e is equal to the waste flow q (Fig 3.1).

$$q = Q_i - Q_e \tag{3.109}$$

A distinction can now be made between <u>soluble</u> and <u>particulate</u> variables. For <u>soluble</u> variables the effluent concentrations and the concentrations in the waste flow are equal to the reactor concentration ($C_e = C$) and Eq (3.11) reduces to:

$$r_{Cr} = (C - C_i)/R_h$$

where

r_{Cr} = reaction rate of the soluble variable C ($mg.\ell^{-1}d^{-1}$)

The <u>soluble</u> variables in the activated sludge process are the concentrations of:

1) Easily biodegradable organic material (S_{bs}) ($mg.COD.\ell^{-1}$)
2) Soluble unbiodegradable organic material (S_u) ($mg.COD.\ell^{-1}$)
3) Soluble unbiodegradable nitrogen (N_u) ($mg.\ N.\ell^{-1}$)
4) Ammoniacal nitrogen (N_a) ($mg.N.\ell^{-1}$)
5) Organic nitrogen (N_o) ($mg.\ N.\ell^{-1}$)
6) Nitrate (N_n) ($mg.\ N.\ell^{-1}$)
7) Alkalinity (Alk) ($mg.C_aCO_3.\ell^{-1}$)

For <u>particulate</u> variables the effluent concentration is zero because the settler removes these variables from the mixed liquor and

recycles these to the reactor. For these variables Eq (3.108) can be written as:

$$dC/dt = (Q_i C_i - q.C)/V + r_{Cr} = 0 \qquad (3.110)$$

With Eqs (3.105) and (3.106), Eq (3.110) reduces to:

$$r_{Cr} = C/R_s - C_i/R_h \qquad (3.111)$$

The <u>particulate</u> variables in the activated sludge process are the concentrations of:

1) Particulate biodegradable material (S_{bp}) ($mg.COD.\ell^{-1}$)
2) Particulate unbiodegradable material (X_i) ($mg.VSS.\ell^{-1}$)
3) Active sludge (X_a) ($mg.VSS.\ell^{-1}$)
4) Endogenous residue (X_e) ($mg.VSS.\ell^{-1}$)
5) Stored biodegradable material (X_s) ($mg.VSS.\ell^{-1}$)
6) Nitrosomonas (X_n) ($mg.VSS.\ell^{-1}$)
7) Volatile solids (X_v) (dependent variable) ($mg.\ VSS.\ell^{-1}$)

By substituting for the reaction rates r_{Cr} for the different variables from Eqs (3.86) to (3.95) from Table 3.1 in Eq (3.109) for soluble variables, and in Eq (3.111) for particulate variables the reaction rates can be directly linked to influent and effluent concentrations, the influent flow, the reactor volume and the sludge age. For the unbiodegradable variables the reaction rates are zero, i.e.

$$(r_{su})_r = (r_{xi})_r = (r_{nu})_r = 0$$

The reaction rates of the thirteen independent variables and of the dependent variable X_v for the completely mixed activated sludge process under steady start conditions are given in Table 3.2. The oxygen uptake rates for carbonaceous energy removal, O_c, and for nitrification, O_n, are expressed in terms of the rates of utilization of organic material and nitrification respectively.

Table 3.2 Reaction rates of the fourteen independent variables and the dependent variable in a steady state single reactor activated sludge process.

$$r_{sbs} = -r_{us} = -(S_{bs}-S_{bsi})/R_h \quad (3.112)$$

$$r_{sbp} = -r_a + P(1-f')r_d = S_{bp}/R_s - S_{bpi}/R_h \quad (3.113)$$

$$r_{xs} = (r_a - r_{up})/P = X_s/R_s \quad (3.114)$$

$$r_{xa} = Y(r_{us} + r_{up}) - r_d = X_a/R_s \quad (3.115)$$

$$r_{xe} = f'.r_d = X_e/R_s \quad (3.116)$$

$$r_{xn} = Y_n r_n - r_d = X_n/R_s \quad (3.117)$$

$$r_{na} = -r_n - Y_h(r_{us} + r_{up}) + r_{co} = (N_a - N_{ai})/R_h \quad (3.118)$$

$$r_{nn} = r_n = (N_n - N_{ni})/R_h \quad (3.119)$$

$$r_{no} = -r_{co} - f_{ns}(r_a - r_{up}) + f_n(1-f')r_d = (N_o - N_{oi})/R_h \quad (3.120)$$

$$r_{alk} = 7{,}14\ r_n + 3{,}57\,(r_{co} - f_n Y_h (r_{us} + r_{up})) \quad (3.121)$$

$$= (Alk - Alk_i)/R_h \quad (3.122)$$

$$r_{su} = 0 = (S_u - S_{ui})/R_h \quad (3.123)$$

$$r_{xi} = 0 = X_i/R_s - X_{ii}/R_h \quad (3.124)$$

$$r_{nu} = 0 = (N_u - N_{ui})/R_h \quad (3.125)$$

$$r_{xv} = r_{sbp}/P + r_{xs} + r_{xa} + r_{xe} = X_v/R_s - X_{ii}/R_h \quad (3.126)$$

$$O_c = (1-PY_h)\ (r_{us}+r_{up}) \quad (3.127)$$

$$O_n = 4{,}57\ r_n \quad (3.128)$$

9.3 Ideal single reactor completely mixed activated sludge process

For the ideal case of complete utilization of biodegradable organic material a simple analytical solution for Eqs (3.112) to (3.128) was presented by Marais and Ekama (1976). This solution will be discussed in this section. In actuality an activated sludge process is a non-ideal system, i.e. not all the influent biodegradable material is utilized; a certain fraction is discharged either with the effluent or with sludge wastage. A numerical solution procedure for the non-ideal activated sludge process is feasible and will be presented in Section 9.4.

9.3.1 Carbonaceous energy removal

As a consequence of the assumption of complete removal of biodegradable material in the ideal activated sludge process: $S_{bs} = S_{bp} = X_s = 0$, the concentrations of the sludge fractions and the oxygen uptake rate for carbonaceous energy removal can be calculated as follows:

a) Active sludge concentration

From Eq (3.112)

$$r_{us} = S_{bsi}/R_h \qquad (3.129)$$

From Eq (3.113)

$$r_a = S_{bpi}/R_h + P(1-f')r_d \qquad (3.130)$$

From Eq (3.114)

$$r_{up} = r_a \qquad (3.131)$$

From Eq (3.115)

$$X_a = R_s(Y_h(r_{us}+r_{up}) - r_d) \qquad (3.132)$$

Substituting for r_{us}, r_{up} and r_d from Eqs (3.78), (3.79), and (3.81) respectively:

$$X_a = Y_h R_s (S_{bsi} + S_{bpi}/R_h + b'_{hT}(1-f')PX_a) - b'_{hT} R_s X_a \quad (3.133)$$

Solving for X_a from Eq (3.133)

$$X_a = Y_h R_s (S_{bsi}+S_{bpi}) / \ (1+b'_{hT} R_s (1-PY_h (1-f')))R_h$$

$$= Y_h R_s \ S_{bi} / [(1+b'_{hT} R_s (1-PY_h (1-f')))R_h] \quad (3.134)$$

The expression in Eq (3.134) takes a simpler form when it is written in terms of the endogenous respiration approach. From Eq (3.35):

$$X_a = Y_h R_s S_{bi} / [(1 + b_{hT} R_s) \ R_h] \quad (3.135)$$

b) Endogenous residue

Substituting for r_d from Eq (3.81) in (3.118):

$$X_e = f' b'_{hT} R_s X_a \quad (3.136)$$

In terms of the endogenous respiration approach from Eq (3.36):

$$X_e = f b_{hT} R_s X_a \quad (3.137)$$

c) Unbiodegradable material

The concentrations of unbiodegradable soluble and particulate organic material can be calculated from Eqs (3.123) and (3.124) respectively:

$$S_u = S_{ui} = f_{us} S_{ti} \quad (3.138)$$

$$X_i = R_s/(R_h X_{ii}) = R_s/(R_h f_{up} S_{ti}) \quad (3.139)$$

d) Total volatile solids concentration

The concentration of volatile solids is now readily calculated as the sum of the concentrations of active sludge, endogenous residue and inert sludge.

$$X_v = X_a + X_e + X_i$$

$$= Y_h R_s S_{bi}(1+fb_{hT}R_s)/(1+b_{hT}R_s) + R_s f_{up} S_{ti}/R_h$$

$$= [(Y_h R_s(1-f_{us}-Pf_{up})(1+fb_{hT}R_s)/(1+b_{hT}R_s) + R_s f_{up}] S_{ti}/R_h \quad (3.140)$$

e) Oxygen uptake rate

The oxygen uptake rate for carbonaceous energy removal can be expressed with Eq (3.96)

$$O_c = (1-PY_h)(r_{us}+r_{up}) \quad (3.141)$$

Substituting for r_{us} and r_{up} from Eqs (3.129) and (3.131) respectively:

$$O_c = (1-PY_h)((S_{bsi}+S_{bpi})/R_h + P(1-f')b'_{hT}X_a) \quad (3.142)$$

From Eq (3.37) the last equation can be rewritten in terms of the endogenous respiration approach

$$O_c = ((1-PY_h)S_{bi}/R_h + P(1-f)b_{hT}Y_h R_s/(1+b_{hT}R_s))S_{bi}/R_h$$

$$= (1-PY_h + P(1-f)b_{hT}Y_h R_s/(1+b_{hT}R_s))(1-f_{us}-Pf_{up})S_{ti}/R_h \quad (3.143)$$

All the independent variables (S_u, S_{bs}, S_{bp}, X_s, X_a, X_e and X_a) and the dependent variables (X_v, O_c) of the process of carbonaceous oxygen removal are now expressed in terms of sewage characteristics (S_{ti}, f_{us}, f_{up}, T), mass parameters of sludge (P, Y_h, f) the endogenous respiration rate constant, b_{hT}, and operational variables (R_s, R_h).

In the discussion above the steady state equations of an ideal completely mixed single reactor activated sludge process were derived in terms of both the death regeneration and the endogenous respiration approaches. Because both gave identical numerical values for the

variables X_a, X_e, X_i and O_c, for reasons of convenience the simplest expressions should be used. These are clearly the expressions based on the endogenous respiration approach; consequently further description of the process will be in terms of this approach. In Table 3.3 the equations defining carbonaceous energy removal in the ideal completely mixed single reactor activated sludge process are listed for convenience. If complete utilization of biodegradable material in the single reactor system is assumed the masses or concentrations of organic sludge and the oxygen uptake rate can be predicted in terms of the influent COD mass and the constants related to the sewage characteristics (f_{us}, f_{up}) and the sludge (P, Y_h, f, b_{hT}); and the sludge, R_s. To determine the values of the constants it is necessary to measure five relevant parameters:

1. Influent COD, S_{ti}.
2. Effluent COD, S_t
3. Total volatile sludge concentration, X_v.
4. COD equivalent of volatile sludge (p-ratio).
5. Oxygen uptake rate for oxidation of organic material.

The last parameter can in fact be measured independently only if no nitrification takes place. However it will be shown in the next section that if nitrification occurs, O_c can be calculated from the difference between the total oxygen uptake rate O_t and the oxygen uptake rate for nitrification O_n; the latter is obtained from the increase in nitrate concentration. If complete utilization of biodegradable material is assumed then the effluent COD concentration, S_t, can be equalled to the unbiodegradable soluble influent COD.

$$S_t = S_u = f_{us} S_{ti} \tag{3.160}$$

Hence the unbiodegradable soluble influent COD fraction, f_{us}, is readily determined as:

$$f_{us} = S_t / S_{ti} \tag{3.161}$$

Table 3.3 Independent and dependent variables associated with carbonaceous energy removal in the ideal completely mixed single reactor activated sludge process.

$$S_u = f_{us} S_{ti} \tag{3.144}$$

$$S_{bs} = 0 \tag{3.145}$$

$$S_{bp} = 0 \tag{3.146}$$

$$X_s = 0 \tag{3.147}$$

$$X_a = Y_h R_s (1-f_{us}-Pf_{up}) S_{ti} . ((1+b_{hT} R_s)) R_h \tag{3.148}$$

$$X_e = f b_{hT} R_s S_{ti} / R_h \tag{3.149}$$

$$X_i = f_{up} R_s S_{ti} / R_h \tag{3.150}$$

$$X_v = ((1+f b_{hT} R_s)(1-f_{us} P f_{up}) / (1+b_{hT} R_s) + f_{up} R_s) S_{ti} / R_h \tag{3.151}$$

$$O_c = [1 - PY_h + P(1-f) b_{hT} Y_h R_s (1-f_{us}-Pf_{up}) / ((1+b_{hT} R_s) R_h)] S_{ti} \tag{3.152}$$

The constant P, the COD:VSS ratio can be determined by doing a COD test on a sludge sample with known organic solids concentration. In principle the remaining unknown constants f_{up}, Y_h, b_{hT} and f can be determined by operating the system at several sludge ages, R_s, and measuring the values of X_v and/or O_c. However, Marais and Ekama (1976) have shown that in practice this procedure does not lead to reliable results, because it is always possible to find pairs of values for b_{hT} and Y_h that give at any particular sludge age almost identical results for X_v and O_c. This difficulty can be overcome only if one of the two constants is measured independently. Marais and Ekama (1976) have shown that it is possible to determine the endogenous respiration rate constant, b_{hT}, independently from Y_h in an aerobic batch digestor by observing the change in oxygen uptake rate for carbonaceous energy removal as a function of time.

From experiments on aerobic digestors (batch tests) Marais and Ekama (1976) determined the endogenous respiration rate constant b_{hT} and the unbiodegradable fraction of active sludge, f, the values being calculated from observations of the carbonaceous oxygen uptake rate. Marais and Ekama hypothesized that the values of b_{hT} and f thus determined were applicable also to activated sludge systems. Then from observations of the concentration of volatile sludge and the carbonaceous oxygen uptake rate in steady state single reactor activated sludge systems the values of Y_h and f_{up} could be determined. (The experiments were carried out at 20°C and at 14°C and sludge ages ranging from 2,5 to 30 days). Table 3.4 shows the numerical values of sewage characteristics (f_{us} and f_{up}), sludge mass parameters (f, f', Y_h, P) and kinetic constants (b_{hT}, b'_{hT}) relevant to the removal of carbonaceous energy in the ideal single reactor activated sludge process under constant flow and load conditions.

In Table 3.4 a f_{us} value of 0,10 instead of the value reported by Marais and Ekama (1976) (0,05 mg COD.mg COD^{-1}) is used because it was found that in the experiments described in the next Chapter the effluent COD concentration was about 10 per cent of the influent concentration, and this value is in accordance with effluent COD reported by other research workers (see Section 3.1.1 of this Chapter).

Table 3.4 Constants relevant to the ideal aerobic activated sludge process after Marais and Ekama (1976)

A) Sewage Characteristics	Raw sewage	Settled sewage
f_{us} (mg COD.mg COD^{-1})	0,1*	0,1*
f_{up} (mg VSS.mg COD^{-1})	0,09	0,02 - 0,05

B) Sludge Characteristics

$$b_{hT} = 0,24(1,029)^{(T-20)} \quad mg\ X_a.mg\ X_a.d^{-1} \qquad (3.162)$$

$$b'_{hT} = 0,62(1,029)^{T-20} \quad mg\ X_a.mg\ X_a.d^{-1} \qquad (3.163)$$

$$f = 0,2\ mg\ X_e.mg\ X_a^{-1} \qquad (3.164)$$

$$f' = 0,08\ mg\ X_e.mg\ X_a^{-1} \qquad (3.165)$$

$$P = 1,48\ mg\ COD.mg\ VSS^{-1} \qquad (3.166)$$

$$Y_h = 0,45\ mg\ VSS.mg\ COD^{-1} \qquad (3.167)$$

* Estimated from experiments described in the present work (Chapter 4).

The hydraulic retention time R_h of the process can be eliminated from the process equations (Eqs (3.144) to (3.152)) by considering the mass of sludge present in the process and the daily mass of influent applied to the process: With $R_h = V/Q$, Eqs (3.144) to (3.152) can be rewritten as:

$$MS_{ti} = Q.S_{ti} \tag{3.153}$$

$$MX_a = V.X_a = YR_s\ (1-f_{us}-Pf_{up})\ MS_{ti}/(1+b_{hT}R_s) \tag{3.154}$$

$$MX_e = V.X_e = fb_{hT}\ R_s MX_a \tag{3.155}$$

$$MX_i = V.X_i = R_s f_{up} MS_{ti} \tag{3.156}$$

$$MX_v = V.X_v = MX_a + MX_e + MX_i \tag{3,157}$$

$$= (YR_s(1+fb_{hT}R_s)(1-f_{us}-Pf_{up})/(1+b_{hT}R_s) - f_{up}R_s)MS_{ti} \tag{3.158}$$

$$MO_c = V.O_c = (1-PY_h + P(1-f)b_{hT}Y_h R_s/(1+b_{hT}R_s))(1-f_{us}-Pf_{up})MS_{ti} \tag{3.159}$$

The mass equations above (Eqs. (3.153) to (3.159)) lead to some interesting concepts discussed below.

9.3.1.1 Sludge Mass

The mass of active sludge generated per unit applied COD per day, m_{xa}, can be calculated from Eqs (3.153) and (3.154).

$$m_{xa} = MX_a/MS_{ti} = (1-f_{us}-P.f_{up})Y_h R_s/(1+b_{hT}R_s). \tag{3.168}$$

Similarly from Eq (1.55)

$$m_{xe} = MX_e/MS_{ti} = f\ b_{hT}R_s.MX_a/MS_{ti} \tag{3.169}$$

And from Eq (1.56)

$$m_{xi} = MX_i/MS_{ti} = f_{up}.R_s. \quad (3.170)$$

Hence

$$m_{xv} = (MX_a+MX_e+MX_i)/MS_{ti}$$

$$= (1-f_{us}-P.f_{up}) (1+f\, b_{hT}R_s/(1+b_{hT}R_s) + f_{up}R_s \quad (3.171)$$

In Fig. 3.5a the values of m_{xa}, m_{xe}, m_{xi} and m_{xv} are shown plotted for raw sewage with f_{us} = 0,1 mg COD. mg COD^{-1} and f_{up} = 0,09 mg VSS/mg COD. Fig 3.5b is a similar plot for settled sewage with f_{us} = 0,1 mg COD.mg COD^{-1} and f_{up} = 0,02 mg VSS.mg COD^{-1}. It may be noted that the sludge mass per unit daily applied COD is very different when f_{up} is decreased from 0,09 to 0,02 mg VSS.mg COD^{-1}. If experimental data for X_v and S_{ti} at different sludge ages are available then the experimental m_{xv} values can be plotted as a function of the sludge age and compared with theoretical curves calculated from Eq (3.171) for different values of f_{up}. Thus by curve fitting the value of f_{up} in a particular sewage source may be estimated. This procedure has been used in the discussion of the work of Sutton *et al* (1979) and Heideman (1979) in Chapter 2.

9.3.1.2 Sludge production.

The daily mass of wasted sludge is a fraction $1/R_s$ of the mass present in the system (Eq (3.105)). The daily mass of sludge produced per unit of daily COD applied therefore is given by:

$$w_{xa} = m_{xa}/R_s = (1-f_{us}-P.f_{up})\, Y_h/(1+b_{hT}R_s) \quad (3.172)$$

$$w_{xe} = m_{xe}/R_s = f\, bR_s.w_{xa} \quad (3.173)$$

$$w_{xi} = m_{xi}/R_s = f_{up} \quad (3.174)$$

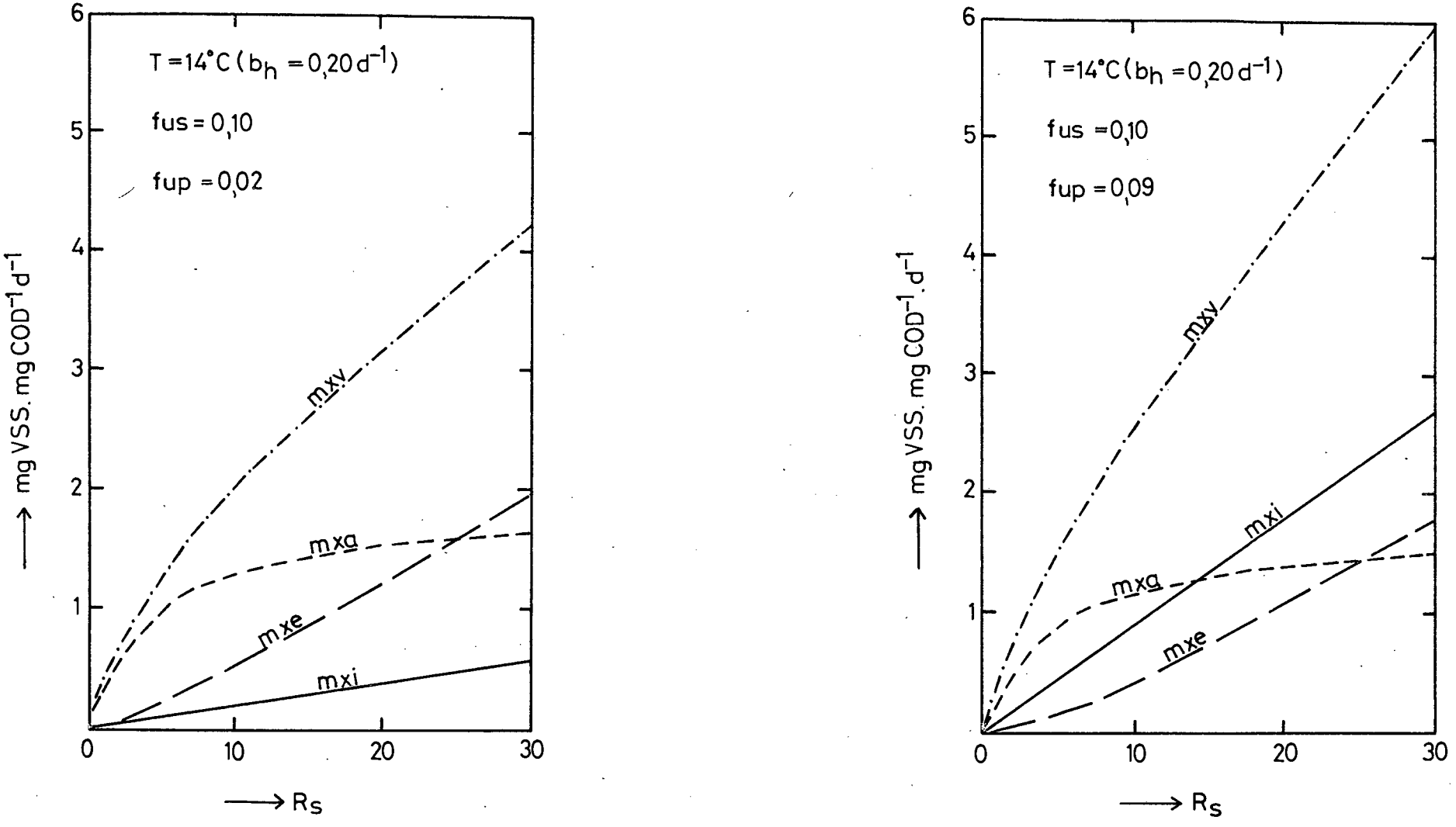

Fig. 3.5 *Masses of active sludge, endogenous residue, inert material and volatile sludge per unit applied COD for settled (f_{up} = 0,02 mg VSS.mgCOD^{-1}, left hand side) and raw sewage (f_{up} = 0,09 mg VSS. mgCOD^{-1}, right hand side) as a function of the sludge age in an aerobic activated sludge process.*

$$w_{xv} = m_{xv}/R_s = (1-f_{up}-P.f_{us})(1+f\ b_{hT}R_s).Y_h/(1+b_{hT}\ R_s) + f_{up} \quad (3.175)$$

In Fig. 3.6 the production of volatile sludge per unit applied COD is shown plotted as a function of sludge age for f_{up} = 0,02 and f_{up} = 0,09 respectively. The fraction of the influent COD that is used for the production of wasted sludge is given by $w_{COD} = P.w_{xv}$. In Fig 3.6, w_{COD} can be read off as a function of the sludge age for raw (f_{us} = 0,1; f_{up} = 0,09) and settled (f_{us} = 0,1 f_{up} = 0,02) sewage (right hand side axis). w_{COD} represents the fraction of the influent COD that is incorporated in the sludge.

9.3.1.3 Nutrient requirements for sludge production.

From Fig 3.6 it is possible not only to determine sludge production per unit applied COD (either as VSS or as COD) but also the required concentrations of the nutrients in the sewage relative to the influent COD ratio. Assuming that sludge has a mass fraction of f_n as nitrogen and f_p as phosphorous then the mass of nitrogen and phosphorous wasted with sludge per unit influent COD is given by respectively:

$$w_N = f_n.w_{xv} \quad (3.176)$$

$$w_P = f_p\ w_{xv} \quad (3.177)$$

With an influent daily mass of $MS_{ti} = Q.S_{ti}$ the required influent daily masses of nutrients nitrogen and phosphorous $MN_s = Q.N_s$ and $MP_s = Q.P_s$ respectively for sludge production are given by:

$$MN_s = Q.N_s = f_n.w_{xv}.Q.S_{ti} \quad (3.178)$$

$$MP_s = Q.P_s = f_p.w_{xv}.Q.S_{ti} \quad (3.179)$$

Hence the removals of nitrogen and phosphorous per litre influent

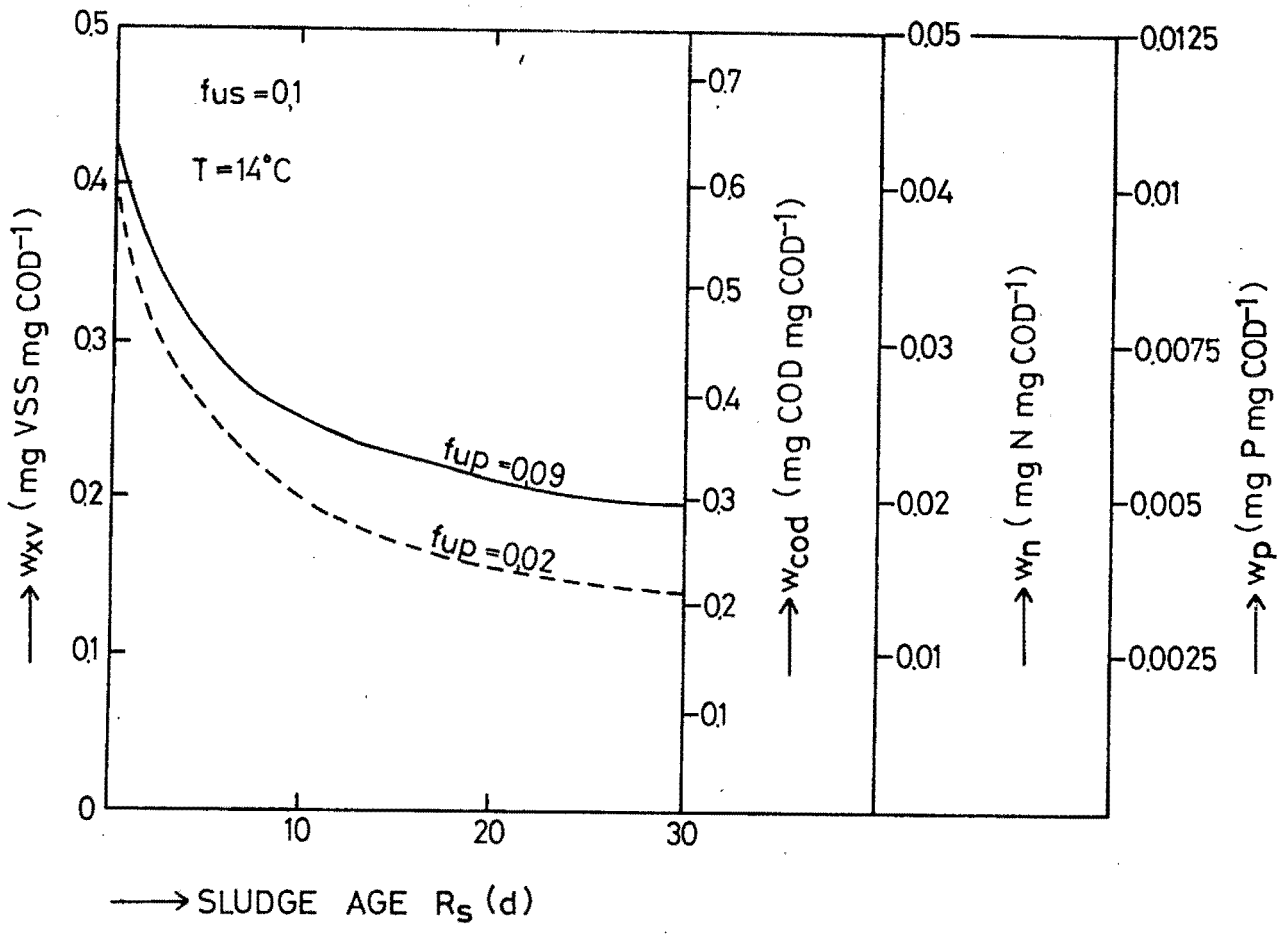

Fig. 3.6 *Sludge production and nutrient requirements (COD, Nitrogen and Phosphorus) for sludge production as a function of sludge age.*

due to sludge production given respectively by

$$N_s = f_n . w_{xv} . S_{ti}$$

$$= f_n [Y_h (1-f_{up}-Pf_{us}) (1+f\ b_{hT} R_s)/(1+b_{hT} R_s)+f_{up}]\ S_{ti} \qquad (3.180)$$

and

$$P_s = f_p . w_{xv} . S_{ti}$$

$$= f_p [. Y_h\ (1-f_{us}-P . f_{up}) (1+f\ b_{hT} R_s)/(1+b_{hT} R_s)+f_{up}]\ S_{ti} \qquad (3.181)$$

Note that in the derivation of these equations it is assumed that the nitrogen and phosphorous mass fractions in the different components of the sludge (X_a, X_e, X_i) are the same: f_n = 0,1 mg N/mg VSS and f_p = 0,025 mg P/mg VSS.

In Fig 3.6 the removal of nitrogen and phosphorous per unit applied COD is indicated as a function of the sludge age for raw (f_{us} = 0,10; f_{up} = 0,09) and settled (f_{us} = 0,10; f_{up} = 0,02) sewage.

9.3.1.4 Oxygen consumption

The oxygen consumption for carbonaceous energy removal in an ideal single reactor activated sludge process is given by Eq (3.159).

$$MO_c = (1-PY_h + P(1-f)\, b_{hT} Y_h R_s/(1+b_{hT}R_s)\,)\; M\, S_{bi} \qquad (3.182)$$

From Eq (3.182) it may be noted that the oxygen consumption for carbonaceous energy removal is composed of two terms related to the synthesis of influent biodegradable material, $MO_{c,syn}$, and endogenous respiration, $MO_{c,e}$ respectively:

$$MO_{c,syn} = (1-PY_h)MS_{bi} \qquad (3.183)$$

$$MO_{c,e} = P(1-f)b_{hT}Y\, R_s/(1+b_{hT}R_s).MS_{bi} \qquad (3.184)$$

The theoretical minimum value for oxygen consumption is obtained for R_s = 0:

$$MO_{c,min} = MO_{c,syn} = (1-PY_h)M\, S_{bi} = 0,33\; M\, S_{bi} \qquad (3.185)$$

The theoretical maximum value is obtained for $R_s \rightarrow \infty$:

$$MO_{c,max} = MO_{c,syn} + P(1-f)b_{hT}Y_h/b_{hT}.MS_{bi}$$

$$= (1-PY_h f)MS_{bi}$$

$$= 0,87\; S_{bi} \qquad (3.186)$$

Hence the oxygen requirement for carbonaceous oxygen removal ranges between a theoretical minimum of 0,33 mg 0 per mg COD utilized for synthesis only and a maximum of 0,87 mg 0per mg COD utilized for

synthesis and complete sludge digestion in the activated sludge system. The mass of oxygen required per unit of influent COD, m_{oc} can readily be calculated from Eq (3.184):

$$m_{oc} = MO_c/MS_{ti}$$

$$= (1-PY_h+P(1-f)b_{hT}Y_hR_s/(1+b_{hT}R_s))\ (1-f_{us}-P.f_{up}) \qquad (3\ 187)$$

From a mass balance on COD it may be noted that

$$f_{us} + P.w_{xv} + m_{oc} = 1$$

Where f_{us} = fraction of influent COD discharged with the effluent

$P.w_{xv}$ = fraction of COD wasted with sludge

m_{oc} = fraction of COD oxidized in the reactor

Thus for any sludge age it is possible to calculate the fraction of the influent that leaves the system unchanged (f_{us}), the fraction that is oxidized (m_{oc}) and the fraction that is wasted as volatile solids ($P.w_{xv}$). In Fig 3.7 the relative proportions are shown plotted as a function of the sludge age for a temperature of 14°C and for raw (f_{up} = 0,09; f_{us} = 0,1) and settled sewage (f_{up} = 0,02; f_{us} = 0,1).

Note that for all the formulae in this section it is assumed that utilization of biodegradable material is complete. This assumption may be true for a long sludge age when the ratio active sludge: applied daily COD is large but for short sludge ages this assumption is no longer justified. The minimum sludge age for near complete utilization of biodegradable organic material depends upon the kinetic constants for utilization and adsorption of organic substrate. The determination of these constants will be discussed in Section 10. With known values of these kinetic constants it is possible to approximate the variables S_{bs}, S_{bp} and X_s in a non ideal activated sludge system without the aid of a computer. This will be discussed In Section 9.4.

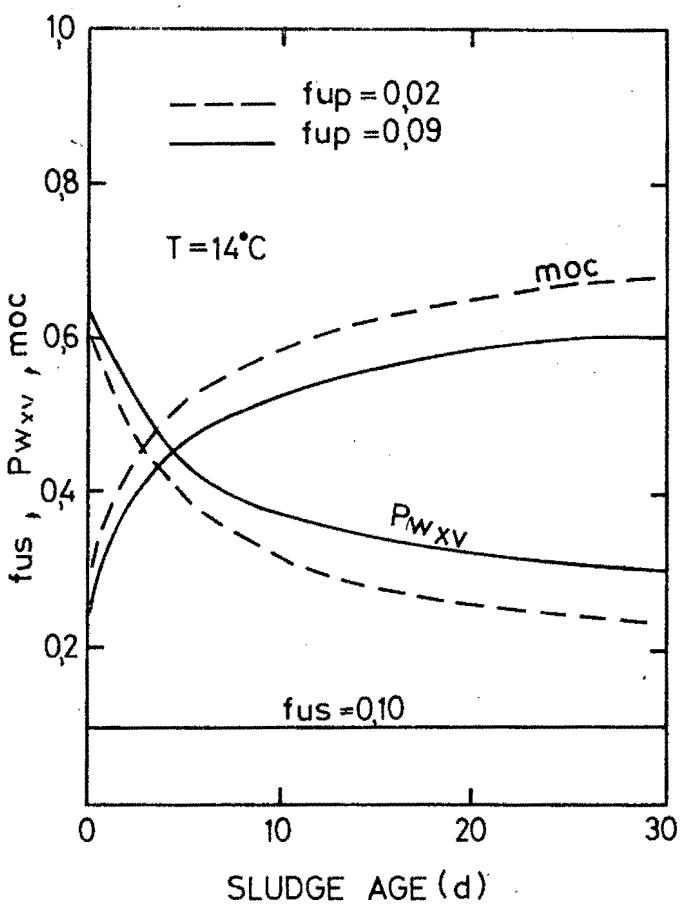

Fig. 3.7 *Fractions of applied COD wasted as sludge ($P.w_{xv}$), discharged with the effluent (f_{us}) and oxidized (m_{oc}) in the aerobic activated sludge process as a function of the sludge age.*

9.3.2 Nitrogenous material in the single reactor completely mixed activated sludge process.

a) Nitrogen requirement for sludge production

In Section 9.3.1.3 it was established that the nitrogen requirements for heterotrophic sludge production can be expressed as:

$$N_s = f_n \left[\frac{Y(1+f\, b_{hT} R_s)}{1+b_{hT} R_s} \cdot (1-f_{us}-P.f_{up}) + f_{up} \cdot \right] S_{ti} \cdot \qquad (3.189)$$

Note that $f_n.f_{up}.S_{ti} = N_{pi}$ (Eq (3.44)), the nitrogen influent fraction associated with the particulate unbiodegradable influent COD fraction, X_{ii}. Hence the nitrogen requirements can be divided into requirements for synthesis of X_a and X_e, N_{syn}, for which ammonia is required and the influent unbiodegradable nitrogen N_{pi} is associated with the unbiodegradable particulate COD concentration in the influent, X_{ii}:

$$N_s = N_{syn} + N_{pi} \quad (3.190)$$

where

$$N_{syn} = f_n.Y_n(1-f_{us}-P.f_{up})(1+f\ b_{hT}R_s)/(1+b_{hT}R_s).S_{ti}. \quad (3.191)$$

and

$$N_{pi} = f_n.f_{us}.S_{ti}. \quad (3.44)$$

It is assumed that the required concentration for production of heterotrophic sludge, i.e., $N_s = N_{syn} + N_{pi}$ is available in the waste water.

The reaction rates and effluent concentrations of the different nitrogenous species, and of Nitrosomonas can now be calculated.

b) Unbiodegradable soluble nitrogen.

If unbiodegradable soluble nitrogen is present its concentration in the effluent is the same as in the influent:

$$N_u = N_{ui} = f_{un}.N_{ti} \quad (3.192)$$

c) Organic nitrogen

From Eq. (3.120) and noting that $r_a = r_{up}$ (Eq (3.131))

$$r_{no} = -r_{co} + f_n\ (1-f').b'_{hT}\ .X_a = (N_o-N_{oi})/R_h \quad (3.193)$$

From Eq (3.84):

$$-K_{rT}N_o.X_a + f_n(1-f')\ b'_{hT}X_a = (N_o-N_{oi})/R_h$$

and

$$N_o = \frac{N_{oi}+f_n(1-f')b'_{hT}X_a}{1+K_{rT}\ X_aR_h}. \quad (3.194)$$

where

$$N_{oi} = (1-f_{na}-f_{un})\, N_{ti} - N_{pi}. \qquad (3.41)$$

Eq (3.194) can be rewritten in terms of b_{hT} and f instead of b_{hT} and f' using Eq (3.37):

$$N_o = \frac{N_{oi} + f_n(1-f)b_{hT} X_a/(1-PY_h)}{1+K_{rT}X_a R_h}. \qquad (3.195)$$

d) Ammoniacal nitrogen.

If no nitrification takes place in the single reactor the effluent ammonia concentration can be calculated as the sum of the influent ammonia concentration plus the ammonia concentration generated internally through ammonification less the nitrogen used for synthesis (deammonification). The extent of ammonification in the reactor is given by the difference between the influent organic nitrogen concentration, N_{oi}, and the effluent concentration, N_o, whereas the organic nitrogen wasted as sludge is equivalent to deammonification of concentration N_{syn} in the influent, so that the <u>nett</u> extent of ammonic to organic nitrogen is:

$$N_{am} = N_{oi} - N_o - N_{syn} \qquad (3.196)$$

Hence the maximum concentration of ammonia, N_p, in the effluent is equal to:

$$N_p = N_{ai} + N_{am} \qquad (3.197)$$

$$= N_{ai} + N_{oi} - N_o - N_{syn}$$

$$= N_{ti} - N_o - N_s - N_u$$

The concentration N_p is termed the <u>nitrification potential</u>; it indicates the concentration of nitrogen in the influent that is available for nitrification. If nitrification does take place the ammonia effluent concentration may be calculated from Eq (3.117).

$$\mu_{nmT} \cdot N_a.X_n./(N_a+K_{nT}) - b_{nT}.X_n = X_n/R_s \quad (3.198)$$

$$N_a = K_{nT}(b_{nT}+1/R_s)/[\mu_{nmT}-(b_{nT}+1/R_s)] \quad (3.199)$$

Obviously if nitrification takes place the effluent ammonia concentration cannot be greater than the maximum value N_p. This implies that there is a minimum sludge age R_{sm} below which no nitrification can take place. The minimum sludge age for nitrification is given by:

$$N_p = K_{nT}(B_{nT}+1/R_{sm})/\ \mu_{nmT}-(b_{nT}+-/R_{sm}) \quad (3.200)$$

Rearranging

$$1/R_{sm} = \mu_{nmT}.(N_p/K_{nT})/(1+N_p/K_{nT})-b_{nT} \quad (3.201)$$

From Chapter 1 (table 1.7) it is known that K_n is small so that $N_p/K_{nT} \ll 1$ and Eq (3.201) approximates to

$$R_{sm} = 1(\mu_{nmT}-b_{nT}) \quad (3.201a)$$

For any sludge age $R_s > R_{sm}$ nitrification will take place and the effluent ammonia concentration is given by Eq (3.199). The extent of nitrification is given by the difference between N_p and N_a. This difference is the extent of nitrification that actually takes place in the system and is termed the nitrification capacity, N_c:

$$N_c = N_p - N_a = N_{ti} - N_u - N_o - N_s - N_a$$

$$= N_{ti} - N_a - N_o - N_s - K_{nT}(b_{nT}+1/R_s)/(\mu_{nmT}-b_{nT}-1/R_s) \quad (3.202)$$

Under steady state conditions with a hydraulic retention time R_h the rate of nitrification is given by:

$$r_n = N_c/R_h = (N_p-N_a)/R_h \quad (3.203)$$

e) Nitrate concentration

With Eqs (3.203) and (3.119)

$$r_n = (N_n - N_{ni})/R_h = (N_p - N_a)/R_h \quad (3.204)$$

$$N_n = N_{ni} + N_p - N_a \quad (3.205)$$

Where

N_{ni} = influent nitrate concentration.

N_n = effluent nitrate concentration.

f) Concentration of Nitrosomonas.

The concentration of Nitrosomonas can be calculated by substituting Eq (3.203) in Eq (3.117):

$$X_n = Y_n R_s (N_p - N_a)/((1+b_{nT} R_s)\ R_h). \quad (3.206)$$

Note that the expression for X_n is completely analogous to the expression for the active sludge concentration, X_a, (Eq (3.135)).

g) Oxygen uptake rate for nitrification

The oxygen uptake rate for nitrification O_n is calculated from Eq (3.127) using again Eq (3.203):

$$O_n = 4{,}57.(N_p - N_a)/R_h \quad (3.207)$$

The oxygen uptake rate for nitrification calculated using Eq (3.207) may be subtracted from the measured total oxygen uptake rate O_t to give the oxygen uptake rate for carbonaceous material removal, O_c:

$$O_c = O_t - O_n = O_t - 4{,}57\ (N_n - N_{ni})/R_h \quad (3.208)$$

In order to determine the sewage characteristics (f_{un}, f_{na}), kinetic constants (K_{rT}, μ_{nmT}, b_{nT}, K_{nT}) and the sludge mass parameter (f_n) relevant to the reactions of nitrogenous material in the activated sludge is concerned, the following tests need to be carried out:

a) Influent TKN, N_{ti} ($mgN.\ell^{-1}$)

b) Influent ammonia, N_{ai} ($mgN.\ell^{-1}$)

c) Influent nitrate, N_{ni} ($mgN.\ell^{-1}$)

d) Effluent TKN, $N_a + N_o + N_u$ ($mgN.\ell^{-1}$)

e) Effluent ammonia, N_a ($mgN.\ell^{-1}$)

f) Effluent nitrate, N_n ($mgN.\ell^{-1}$)

g) Mass fraction of nitrogen in organic sludge f_n ($mgN.mgVSS^{-1}$)

From experimental observations Marais and Ekama (1976) found that the unbiodegradable soluble influent TKN fraction N_{ui} is negligibly small in domestic wastes. Accepting $N_{ui} = 0$ the remaining influent TKN fractions N_{ai}, N_{oi} and N_{pi} can be determined from the influent TKN and ammonia fractions and the value for f_{up} (Eqs (3.210), (3.211) and (3.212)).

The values of the variables relevant to the reactions of nitrogenous material in a single reactor completely mixed ideal activated sludge process under constant flow and load conditions are summarized in Table 3.5

In Table 3.6 are the numerical values of the sewage characteristics (f_{na} and f_{un}), the mass parameters for autotrophic sludge (Y_n) and the kinetic constants (μ_{nmT}, b_{nT} ^{m} K_{nT} and K_T) for the reactions of nitrogenous species as reported by Marais and Ekama (1976).

Due to the fact that the half saturation value is so small, nitrification under steady state process is always virtually complete when the reaction takes place at all. In Fig 3.8 the concentrations of the effluent nitrogen species are shown plotted as a function of the sludge age for raw sewage (f_{us} = 0,1 mg COD.mg COD^{-1}; f_{up} = 0,09 mg VSS. mg COD^{-1}) with a N_{ti}/S_{ti} ratio of 50/500 mg N.mg COD and with f_{un} = 0,0 and f_{na} = 0,75, a temperature of T = 14°C and a μ_{nm14} value of 0,25 mg X_n.mg X_n^{-1}.d^{-1} are assumed. All other constants have their values listed in Table 3.6. Note that although all the nitrogenous species are soluble, theoretically all the effluent concentrations are

Table 3.5 Effluent concentrations of nitrogenous species in the single reactor completely mixed ideal activated sludge process under constant flow and load conditions.

$$N_u = N_{ui} = f_{un} N_{ti} \tag{3.209}$$

$$N_{pi} = f_n . f_{up} S_{ti} \tag{3.210}$$

$$N_{oi} = (1-f_{na}) N_{ti} - N_{ui} - N_{pi} \tag{3.211}$$

$$N_{ai} = f_{na} . N_{ti} \tag{3.212}$$

$$N_{syn} = f_n [Y_h (1+f b_{hT} R_s)(1-f_{us}-P.f_{up})/(1+b_{hT} R_s)] S_{ti} \tag{3.213}$$

$$N_o = (N_{oi} + f_n (1-f) b_{hT} X_a / (1-PY_h))/(1+k_{rT} X_a R_h) \tag{3.214}$$

$$N_{am} = N_{oi} - N_o - N_s \tag{3.215}$$

$$N_p = N_{ai} + N_{oi} - N_o - N_{syn} \tag{3.216}$$

$$= N_{ti} - N_u - N_o - N_s$$

$$N_a = K_{nT}(b_{nT}+1/R_s)/(\mu_{nmT}-b_{nT}-1/R_s) \qquad (N_a \leqslant N_p) \tag{3.217}$$

$$N_n = N_{ni} + N_p - N_a \tag{3.218}$$

$$X_n = Y_n R_s (N_p-N_a)/((1+b_{nT} R_s) R_h) \tag{3.219}$$

$$O_n = 4{,}57.(N_p-N_a) / R_h \tag{3.220}$$

Table 3.6 Sewage characteristics, sludge parameters and kinetic constants relevant to the reactions of nitrogenous species in the completely mixed single reactor activated sludge system under steady state conditions.

Sewage characteristics:	Raw sew.	Settled sew.
f_{na}	0,65 - 0,75	0,75 - 0,85
f_{un}	0,0	0,0

Kinetic constants and mass parameters

$$b_{nT} = 0{,}04.\ (1.029)^{T-20} \quad \text{mg } X_n.\text{mg } X_n.d^{-1} \qquad (3.221)$$

$$K_{nT} = 1{,}0\ (1.123)^{T-20} \quad \text{mg } NH_3\text{-N}.d^{-1} \qquad (3.222)$$

$$K_{rT} = 0{,}023\ (1{,}10)^{T-20} \quad \ell.\text{mg } X_a^{-1}.d^{-1} \qquad (3.223)$$

$$Y_n = 0{,}10 \quad \text{mg } X_n.\text{mg } NH_3\text{-N}.d^{-1} \qquad (3.224)$$

$$\mu_{nmT} = (0{,}33 \text{ to } 0{,}65) \quad \text{mg } X_n.\text{mg } X_n^{-1}.d^{-1} \qquad (3.225)$$

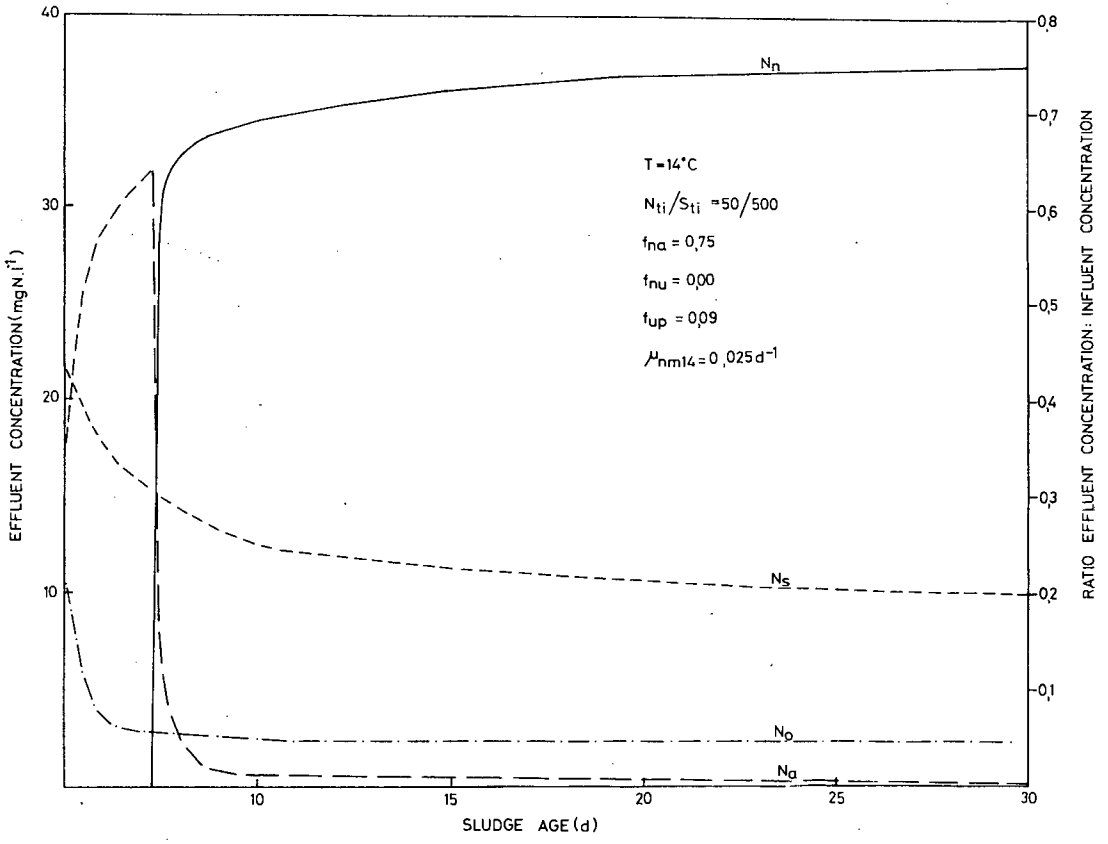

Fig. 3.8 *Concentrations of effluent nitrogen species (Organic nitrogen, N_o, Ammoniacal nitrogen, N_a, and Nitrate nitrogen, N_n) from an aerobic activated sludge process as a function of the sludge age.*

independent on the value of the hydraulic retention time.

9.3.2.1 Determination of nitrification rate constants

In Chapter 2 (Figs 2a, b and c) it has been shown that nitrification behaviour in activated sludge systems is determined principally by the value of the maximum specific growth rate for nitrifiers, μ_{nmT}; the values of b_{nT} and K_{nT} have little influence and exact knowledge of their respective numerical values is not important. Based on reported values Marais and Ekama accepted $b_{nT} = 0,04\ (1,029)^{T-20}$ and $K_{nT} = 1,0\ (1,123)^{T-20}$ and these values were shown to give satisfactory results for different sewages.

In principle it should be possible to determine the value of μ_{nmT} by observing the effluent ammonia and or nitrate concentration of a nitrifying activated sludge system and using Eq (3.199). The difficulty here is that for all μ_{nmT} at sludge ages significantly longer than the minimum for nitrification, the nitrification reaction is virtually complete and the effluent ammonia concentration changes only marginally with sludge age. Consequently μ_{nmT} cannot be determined reliably by this procedure; different μ_{nmT} values will give virtually the same ammonia effluent concentration. One may attempt to assess the value of μ_{nmT} by determining the minimum sludge age for nitrification and using Eq (3.201). However, this method is very tedious requiring gradual small reductions of sludge age over a long period to find R_{sm} from the nitrification behaviour.

A superior approach, to determine the value of μ_{nmT} is to use a series reactor system under constant or cyclic flow and load inputs. The constant is then determined by calibration, calculating theoretical effluent ammonia-time profiles in each reactor of the process for different values of μ_{nmT}, until a value is found that gives close correspondence between predicted and observed effluent ammonia-time prodiles. Theoretically the series reactor system under cyclic flow and load conditions should give the most accurate assessment of μ_{nmT}. In practice however, this procedure has not fulfilled expectations, principally because the response of the series reactor system is very sensitive to hydraulic flow conditions imposed on the various reactor;

slight errors in the recycle rates and/or cyclic influent flow rates give rise to widely changing responses, and a precise estimate of μ_{nmT} is very difficult.

The hydraulic effect is the principle cause of the poor results. To eliminate the hydraulic effect, another experimental procedure to determine μ_{nmT} will be proposed in Chapter 4. The constant is determined in a single reactor process under constant flow and load conditions, the reactor being placed alternately in an anoxic and in an aerobic environment.

9.3.2.2 Determination of the conversion (ammonification) rate constant

The kinetic constant for conversion of organic biodegradable nitrogen to ammonia, K_{rT}, can be calculated from observations of the influent and effluent organic nitrogen concentrations under constant flow and load conditions by using Eq (3.195). However, as in nitrification the problem arises that the conversion reaction is largely complete and that estimates of K_{rT} based on this method yield doubtful results. Hence, also for determination of K_{rT} it is preferable to have a series reactor system operated under cyclic or constant flow and load inputs. From simulation of dynamic processes Dold, Ekama and Marais (1980) determined the value of the conversion rate constant as $K_{rT} = 0{,}023\ (1{,}10)^{T-20}$.

9.3.3 Alkalinity in the single reactor completely mixed activated sludge system

Once the concentrations of the nitrogenous species have been determined, the effect of the reactions of nitrogenous species on the mixed liquor alkalinity can readily be calculated. From Eq (3.121):

$$(Alk-Alk_i)/R_h = -7{,}14r_n + 3{,}57r_{am} \qquad (3.226)$$

where

Alk = effluent alkalinity

Alk_i = influent alkalinity

From Eqs (3.203) and (3.196) respectively:

$$r_n = (N_p - N_a)/R_h = (N_{ti} - N_{syn} - N_{pi} - N_u - N_o - N_a)/R_h \quad (3.227)$$

$$r_{am} = (N_{oi} - N_o - N_{syn})/R_h = (N_{ti} - N_{ai} - N_u - N_{pi} - N_o - N_{syn})/R_h \quad (3.228)$$

Hence:

$$Alk = Alk_i - 7{,}14(N_p - N_a) + 3{,}57\,(N_{oi} - N_o - N_{syn})$$

$$= Alk_i - 3{,}57\,(N_{ti} + N_{ai} - N_o - N_s - N_u - 2N_a) \quad (3.229)$$

In the treatment of many waste waters the concentration of influent organic nitrogen, N_{oi}, is about the same as the sum of the concentration of effluent organic nitrogen, N_o, and the nitrogen wasted with synthesized sludge, N_{syn}. Hence the rate of ammonification (Eq (3.228)) usually is very small and the effect of ammonification on the alkalinity is insignificant compared to the effect of nitrification. The reduction of alkalinity due to nitrification is very considerable and often it is necessary to add alkalinity (usually in the form of lime) to the influent wastewater in order to keep the alkalinity in the process above about 35 ppm $CaCO_3$, a minimum value required for stable operation as shown earlier (Fig. 3.4). The change in pH between influent and effluent depends on the change of alkalinity (i.e. on the rates of nitrification and of ammonification) and the change in acidity, the latter being brought about by a possible change in the concentration of dissolved carbon dioxide. The effect of changes in alkalinity and acidity upon the pH can be read off from a diagram like Fig. 3.4. To use Fig. 3.4 it is necessary to know the concentration of dissolved CO_2.

9.4 The non-ideal single reactor steady state activated sludge process

In Section 9.3 all the equations were based on the assumption that utilization of biodegradable material is complete in the reactor. In reality a certain fraction of the biodegradable material is not utilized in the process, i.e. the process is non-ideal. An estimation of the concentration of the biodegradable variables S_{bs}, S_{bp} and X_s in a steady state, completely mixed, single reactor is possible if the kinetic constants describing the utilization of biodegradable material are known. If the utilization of biodegradable influent material is incomplete then a certain part, say S_b, of the influent biodegradable COD concentration S_{bi}, is not utilized by the active sludge. The unutilized biodegradable material leaves the system either in soluble form (i.e. as S_{bs}) or in particulate form (i.e. as S_{bp} or X_s). Soluble biodegradable COD is discharged by both the effluent and the sludge waste flow but the particulate biodegradable COD can leave the system only via the sludge wastage flow. With concentrations S_{bs}, S_{bp} and X_s for easily biodegradable, particulate biodegradable (enmeshed) and stored material respectively in the reactor, the daily mass of biodegradable COD discharged from the system is:

$$MS_b = Q.S_{bs} + q.S_{bp} + P.q.X_s \qquad (3.230)$$

where

MS_b = daily mass of biodegradable material discharged (mg COD.d^{-1})

Q = influent flow

= effluent flow + sludge wastage flow ($\ell.d^{-1}$)

q = sludge wastage flow ($\ell.d^{-1}$)

Hence the daily mass of biodegradable material introduced into the system but not utilized is given by:

$$Q.S_b = MS_b = Q.S_{bs} + q\,S_{bp} + P.q.X_s \qquad (3.231)$$

where

S_b = influent biodegradable COD concentration that is not utilized in the system (mg COD.ℓ^{-1})

From Eqs. (3.105) and (3.106), dividing by Q:

$$S_b = S_{bs} + (R_h/R_s)\ (S_{bp} + P.X_s) \tag{3.232}$$

It may be noted that generally $(R_h/R_s) << 1$ so that in order to build up and maintain significant concentrations of S_{bp} and X_s only a small concentration of influent biodegradable material is required for this purpose. The utilization of easily biodegradable material is an extremely rapid process and normally utilization of influent easily biodegradable material is virtually complete; the non-ideal behaviour of the activated sludge process is almost exclusively due to incomplete utilization of influent particulate biodegradable material.

If a concentration S_b of the influent biodegradable material, S_{bi}, is not utilized then, effectively, the influent biodegradable COD concentration available for sludge growth is equal to:

$$S'_{bi} = S_{bi} - S_b \tag{3.234}$$

Hence the active sludge concentration for the non-ideal activated sludge process will be given by Eq (3.148):

$$X_a = Y_h R_s (S_{bi}-S_b)/((1+b_{hT}R_s)R_h) \tag{3.235}$$

In order to calculate the concentrations of easily biodegradable, (S_{bs}), slowly biodegradable, (S_{bp}), and stored (X_s) material in the reactor and hence with Eq (3.228) the value of S_b, the expressions derived in Table 3.2 may be used, i.e.

$$r_{sbs} = -r_{us} = -K_{msT}S_{bs}(S_{bs}+k_{ssT}) = (S_{bs}-S_{bsi})/R_h \tag{3.112}$$

$$r_{sbp} = -r_a + P(1-f')b'_{hT}r_d = S_{bp}/R_s - S_{bpi}/R_h \tag{3.113}$$

$$r_{xs} = (r_a-r_{up})/P = X_s/R_s \tag{3.114}$$

Rearranging expressions for S_{bs}, S_{bp} and X_s:

$$S_{bs} = (S_{bsi}-K_{ssT}-K_{msT}R_h X_a) + (S_{bsi}-K_{ssT}-K_{msT}R_h X_a)^2 + 4S_{bsi}K_{ss})/2 \quad (3.236)$$

$$S_{bp} = (S_{bpi}/R_h+P(1-f')b_{hT}'X_a)/(1/R_s+K_{vT}X_a(F_{ma}-X_s/X_a)) \quad (3.237)$$

$$X_s = (-C_2 - \sqrt{(C_2^2 - 4C_1C_3)})/(2C_1) \quad (3.238)$$

where

$$C_1 = (P/R_s+K_{vT}S_{bp})/X_a$$

$$C_2 = K_{vT}S_{bp}(1-K_{spT}/P) - K_{mpT} - K_{spT}/R_s$$

$$C_3 = K_{vT}S_{bp}K_{spT}X_a/P$$

Equations (3.236) to (3.238) do not allow direct calculation of the concentrations S_{bs}, S_{bp} and X_s because the X_a appears in these equations and is itself affected by the efficiency of utilization of biodegradable influent material, (see Eq (3.235)). However, the solution can be found by an iterative calculation procedure using Eq (3.235) to (3.238). The procedure can be summarised as follows:

1. To initiate the calculation procedure assume ideal behaviour i.e. $S_{bs} = S_{bp} = X_s = S_b = 0$, and cauculate X_a from Eq (3.235)
2. With the value of X_a thus obtained, calculate S_{bs} and S_{bp} from Eqs (3.236) and (3.237) respectively. Using the calculated value for S_{bp} calculate X_s from Eq (3.238) and S_b from Eq. (3.233). Calculate a new value for X_a from Eq (3.235) using the calculated value for S_b,
3. With the new value of X_a again carry out the calculations in Step (2) to calculate new values for S_{bs}, S_{bp}, X_s, S_b and X_a. Repeat this iterative procedure until the variables S_{bs}, S_{bp}, X_s, S_b and X_a of two successive iterations do not differ more than a set maximum percentage (for example 0,1 per cent); usually 3 or 4 iterations are sufficient.

Once the values of S_{bs}, S_{bp}, S_x and X_a have been determined the values of other variables affected by incomplete organic substrate utilization can be calculated exactly, i.e.

$$X_e = fb_{hT}R_sYR_s/((1+b_{hT}R_s)R_h)(S_{bi}-S_b) \quad (3.239)$$

$$N_{syn} = (f_{ns}(S_{bp}+PX_s) + f_n(X_a+X_e+X_i))R_h/R_s \quad (3.240)$$

$$O_c = (1-PY_h + P(1-f)b_{hT}YR_s/(1+b_{hT}R_s))(S_{bi}-S_b) \quad (3.241)$$

Calculations were carried out to determine the fraction of biodegradable influent material not utilized as a function of sewage characteristics, kinetic constants and operational conditions. The values for the kinetic constants were chosen over a range with average values equal to those proposed by Dold, Ekama and Marais (see Table 3.7, Section 10). In Fig 3.9a the fraction of unutilized biodegradable material in a completely mixed activated sludge process is shown plotted as a function of the sludge age for different values of K_{mpT} for a temperature T = 14°C and a ratio (easily biodegradable)/(total biodegradable) influent material of f_{ca} = 0,24 mg COD.mg COD^{-1}. The values of all the other kinetic constants are in accordance with Table 3.7, Section 10. Figure 3.9b shows a similar plot for T = 20°C.

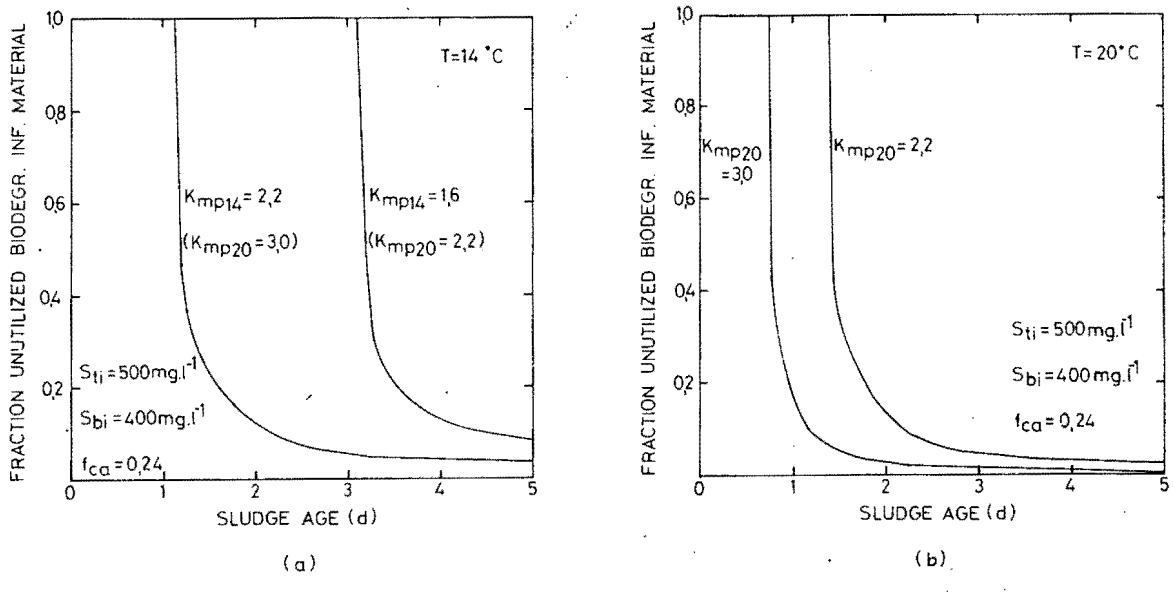

Fig. 3.9 *Fraction of not utilized biodegradable influent material as a function of the sludge age for different K_{mpT} values at 14°C (Fig. 3.9a left hand side) and at 20°C (Fig. 3.9b right hand side).*

From Figs 3.9a and b the following conclusions may be made:

1. The concentration of easily biodegradable material, S_{bs}, in the effluent is always very small, i.e. the utilization of easily biodegradable material is practically complete down to sludge ages of less than 1 day.

2. The concentration of slowly biodegradable material, S_{bp}, in the sludge is only significant if the concentration of stored material X_s is very high, i.e. if a considerable fraction of the "active sites" for the slowly biodegradable material are already occupied.

3. For specified values of the kinetic constants K_{msT}, K_{ssT}, K_{mpT}, K_{spT}, and K_{vT} the fraction of unutilized biodegradable material theoretically depends upon the sludge age but is independent of the hydraulic retention time (See Eqs (3.236) to (3.238)).

4. The constant that most influences the efficiency of utilization of biodegradable material is the substrate utilization constant for slowly biodegradable material, K_{mpT}. Other kinetic constants only have a marginal effect upon the efficiency of the utilization of biodegradable material.

The value of k_{mp} is influenced not only by the temperature of the mixed liquor but also by sewage characteristics. A large proportion of industrial wastes in a sewage stream tends to cause a reduction of the K_{mp} value. The lowest value for K_{mp} determined in Chapter 4 was about K_{mp} = 2,2 mg COD.mg $X_a^{-1}.d^{-1}$ at 20°C. Sehayek and Marais (1981) measured a minimum K_{mpT} value of about 1,6 mg COD.mg $X_a^{-1}.d^{-1}$ at 14°C. If the Arrhenius temperature dependency coefficient of θ = 1,06 proposed by Dold *et al* (see Table 3.7 is accepted) then these values are in good agreement with each other ($K_{mp14} = 0,7\ K_{mp20}$).

Figures 3.9 a and b show that the efficiency of utilization of biodegradable material in the range $0 < R_s < 4$ days strongly depends upon the other K_{mpT} value: At 20°C the lowest measured K_{mp} value is K_{mp20} = 2,2 which require a sludge age of R_s = 2,2 days for 90 per cent efficiency of utilization of biodegradable influent COD; at 14°C for K_{mp14} = 1,6 mg COD.mg $X_a^{-1}.d^{-1}$ a sludge age of R_s = 4,5 days is required for 90 per cent efficiency.

From Fig. 3.9 a it would appear that it is not unreasonable to assume that utilization of biodegradable organic material is "complete" in an aerobic activated sludge process if the sludge age is longer than about 4-5 days, even under unfavourable conditions of a temperature of $T = 14^{o}C$ and a constant for utilization of slowly biodegradable $K_{mp14} = 1{,}6$ mg COD.mg $X_a^{-1}.d^{-1}$. For more favourable values of T and K_{mpT} the sludge age for virtually complete utilization of biodegradable material is significantly less than 4 days (see Fig 3.9 b).

With decreasing sludge age the fraction of biodegradable material that is not utilized in the process increases but this does not necessarily imply that the effluent quality will be poor: The easily biodegradable material (which is soluble) may still be utilized virtually completely whereas particulate material is enmeshed in and stored material is adsorbed on the organism mass i.e. these are part of the solid phase and do not appear in the effluent.

Only under very extreme conditions the effluent quality seems to deteriorate significantly: Alexander and Marais (1979) found that in the contact reactor of a contact-stabilization process the effluent COD (both filtered and unfiltered) increased considerably when the contact time was very short and/or when the feed rate to the contact reactor was very high. Insofar as the increase of filtered effluent COD is concerned, this can possibly be attributed to a reduced efficiency of easily biodegradable material. This reduced efficiency may be explained by assuming that under very high loading the utilization rates of easily and slowly biodegradable material are no longer independent but that interaction of the two processes occurs. This would not appear to be unreasonable, as it is likely that the same "active sites" are used for synthesis of both stored and easily biodegradable soluble material. In the contact reactor under high loading a large fraction of the active sites is occupied by stored material and reduces the accessability for both easily and slowly biodegradable material to the active cells. Hence a reduction in the rate of utilization of easily biodegradabale material and a consequential increase of the filtered effluent COD are to be expected. The fact that in the high load contact - stabilization experiments there was a high concentration of filterable COD in the effluent indicates that the enmeshment of particulate biodegradable material was incomplete, possibly due to the very short retention time in the contact reactor and the low sludge concentration.

10. GENERAL SOLUTION FOR THE AEROBIC ACTIVATED SLUDGE PROCESS

10.1 General formulations

In Section 9.3 it was shown that an analytical solution for activated sludge kinetics is possible in the special case of a completely mixed single reactor process under constant flow and load conditions if complete utilization of biodegradable organic influent material takes place. In Section 9.4 an iterative calculation procedure was presented that gives the solution for such a process if the utilization of influent biodegradable material is not complete.

When the process configuration is made up of a series of reactors, or the flow and load inputs to the system is daily cyclic it is no longer possible to obtain analytical solutions to the process response. The only way possible is to obtain a numerical solution of the set of differential equations defining the process under the imposed inputs for each reactor in the system duly interlinked with the other reactors via the recycles, by assuming each reactor completely mixed. Numerical solutions are feasible only with the aid of a computer.

The simplest numerical solution procedure is to write the differential quotients indicating the rates of change of the concentration of variables, C, in a particular reactor, j, in the discrete finite difference form:

$$r_{Cj} = (dC/dt)_j = (\Delta C/\Delta t)_j \qquad (3.247)$$

where

r_{cj} = rate of change in the concentration of the variable C in reactor j.

C = response variable

ΔC = change in concentration of variable C in reactor j over a short interval t.

Δt = reactor number.

From the general equation for r_{Cj} (Eq (3.100)) the rate of change of C in reactor j depends upon both reaction and hydraulic effects:

$$C_{j,t+\Delta t} - C_{j,t} = (r_{Cjr} + r_{Cjh})\Delta t \qquad (3.248)$$

where

$C_{j,t}$ = value of the variable and time t in reactor j

$C_{j,t+\Delta t}$ = value of variable at time t+ t in reactor j.

r_{Cjr} = rate of change of variable C at time t fue to reaction

r_{Cjh} = rate of change of variable C at time t due to hydraulic effects.

Expressions for r_{Cjr} are given in Eqs. (3.86) to (3.95) and for r_{Cjh} in Eq. (3.103).

The concentrations of the variables, as a function of time are determined by successive integration steps over time intervals Δt. Ekama and Marais (1978) have shown that the discrete finite difference step prediction method expressed by Eq (3.248) is as useful and efficient as any of the more sophisticated integration methods as for example the prediction-correction method of Runge-Kutt .

The main difficulties in applying the solution procedure are (1) to determine the maximum step length for a stable, convergent solution and (2) to estimate the starting values for the variables.

(1) Ekama and Marais (1978) found that the maximum step length for the utilization of slowly biodegradable material is 6 minutes and for the utilization of easily biodegradable material and nitrification 1,5 minutes. The difference arises because the ammonia concentration in a nitrifying activated sludge plant usually is very small and so is the half saturation constant for nitrification. Consequently a small change in the ammonia concentration is reflected in a considerable change in the rate of nitrification; a step length of longer than about 1,5 minutes tends to give rise to an unstable, oscillating response. The same consideration

applies to the utilization of easily biodegradable material, the step length also needs to be limited to 1,5 minutes.

There is no way in which the validity of a numerical solution can be established out of hand. However, one criterion that always must be obeyed is that if a solution is obtained with a particular integration time step length Δt the same solution must again be obtained if the integration time step length is reduced to a value smaller than Δt. Ekama and Marais (1978) have found this to be the case for maximum step lengths of 1,5 minutes for nitrification and utilization of easily biodegradable material and 6 minutes for all the other reactions occurring in the activated sludge process.

(2) Insofar as the selection of the starting values is concerned this depends upon the purpose for which the process solutions are required. In this investigation there were two objectives: (1) to obtain dynamic steady state solutions for processes under cyclic inputs of flow and load and (2) to obtain estimates of specific reaction constants by calibration of simulated data against observed data. These two objectives lead to different solution procedures that will now be discussed.

10.2 Solution for dynamic steady state

For the solution procedure for dynamic steady state, proposed by Ekama and Marais (1978), to be applicable it is necessary that the sewage input flow and load is either constant or of a cyclic nature. The starting values of the variables in each reactor of a multireactor system were obtained by assuming constant flow and load conditions equal to the mean of the cyclic flow and load (COD, TKN, NO_3 and Alkalinity) for a single reactor with a volume equal to that of the sum of the reactors in the system and for the same operational conditions (sludge age, temperature). The steady state solutions for the different variables were determined using the equations for the ideal single reactor presented in Section 9.3. The ideal solution does not give an estimate of the stored

organic material concentration, X_s. Ekama and Marais (1978), in order to obtain some estimate of X_s, used the monosubstrate theory presented earlier by Marais and Ekama (1976) to calculate a theoretical effluent biodegradable COD concentration and equated this concentration to X_s.

The solution procedure for the system under cyclic flow and load conditions then proceeds as follows:

(1) Using the starting values calculate the values of the variables as a function of time over a period of one cycle (i.e. usually one day) for each reactor taking into account recycle flows and concentrations to and from the reactors.

(2) Store the calculated values of the variables over the day.

(3) The final values of the variables (i.e. the values after a complete cycle) serve as starting values for a new iteration and the concentrations in the recycles now vary in accordance with the stored concentrations from the previous cycle.

(4) Compare the values of the variables in the latter iteration with those of the previous one. If the difference between any corresponding pair of variables in any of the reactors at any moment of the cycle is greater than a set maximum (for example 0,1 per cent) then the old stored values are replaced by the values from the last iteration and another iteration over a cycle period is carried out from Step (3).

(5) Steps (3) and (4) are repeated until the difference between any corresponding pair of variables in two successive iterations in each reactor and at any moment of the cycle is smaller than the set maximum difference. When this occurs the values of the variables of the last iteration are accepted as the numerical solutions of the differential equations for the dynamic steady state.

The solution procedure of Ekama and Marais (1978) set out above does not always give satisfactory results: The situation may arise that the numerical solution of a variable is still very different from the "time" solution, but due to slow convergence, the difference between two successive

iterations may be smaller than the maximum set for the comparison criterion. In this case an invalid numerical solution is obtained. The remedy for such behaviour is to impose a stricter comparison criterion but this may cause that the number of required iterations becomes excessive.
In seeking more general and rapidly converging solutions, the reason for bad convergence or an excessive number of iterations became apparent - the estimations of the starting values of some of the variables differed too greatly from the "true" values in the reactors under cyclic flow and load conditions. From large numbers of simulations it became apparent that the estimation of the starting values of two variables was particularly poor; those of the ammonia, N_a, and the stored organic material, X_s, concentrations.

Insofar as the estimation of the ammonia concentration was concerned, the solution process was significantly improved by taking due account of the reduction of nitrification efficiency with increasing severity of the cyclic loading pattern. A better estimate for the ammonia concentration was obtained by multiplying the steady state solution by a factor F defined as follows:

$$F = (A_f + A_{TKN}) / R_h + 1 \qquad (3.249)$$

where

F = correction factor for the ammonia concentration under cyclic flow and load conditions.

A_f = fractional amplitude of the influent flow.

A_{TKN} = fractional amplitude of the influent TKN.

R_h = mean hydraulic retention time.

The fractional amplitude of cyclic flow or concentration is quite easily determined in the case of a sinoidal pattern.

$$A = (V_{max} - V_{min}) / (2V_{av})$$

where

A = fractional amplitude of flow or concentration.

V_{max} = maximum flow or concentration.

V_{min} = minimum flow or concentration.

V_{av} = average flow or concentration.

For a general cyclic pattern of flow or concentration, the fractional amplitude may be determined approximately as follows: (1) Determine the mean value V'_{av} of the flow or concentration, (2) Determine the value V'_{max} of the flow or concentration values above the mean V'_{av} over the whole cycle, (3) Determine the value V'_{min} of the flow or concentration values below the mean value V'_{av} over the whole cycle. The fractional amplitude of the flow or concentration is now given approximately by:

$$A = (V'_{max} - V'_{min})/(V'_{av})$$

With regard to the stored material concentration an improved starting value was found from the application of the nonideal single reactor model using Eq (3.238). Using this starting value for X_s, not only improved the convergence of X_s to the correct value, but also significantly reduced the number of iterations required to find the solution.

The modifications to the starting values, discussed above, were found to be essential in obtaining solutions for systems in which anoxic and anaerobic reactors are included.

Once the adaptations for the starting values for the concentrations of ammonia and stored organic material were instituted, it was tested whether the solution was "well behaved" or not. This was done as follows: The starting values obtained from the steady state solution were either decreased or increased by 10 per cent and these new values were inserted into the iterative calculation procedure; in both cases the numerical solutions converged to the same values as those obtained using the steady state solution but the number of iterations increased significantly.

Subsequent to the development of the bisubstrate-active site-death

regeneration model the original program developed by Ekama and Marais (1978) was modified in association with Dold, Ekama and Marais (1981) to include these new aspects as well as the adaptations discussed above. The modified program is listed in Appendix A.

The main differences between the modified and the original program are:

(1) The theory of adsorption with associated oxygen consumption described by Ekama and Marais is discarded and the bisubstrate theory (i.e. the division of the biodegradable organic material in easily and slowly biodegradable material with independent utilization rates) is introduced.

(2) The Monod function for slowly biodegradable particulate material is replaced by an "active site" approach (Dold, Ekama and Marais, 1980).

(3) The endogenous respiration approach is replaced by the death regeneration approach.

(4) Ammonification is explicitly introduced as a reaction mechanism.

(5) Alkalinity is introduced as an additional variable.

(6) Estimation of the improved starting values for the ammonia and stored biodegradable material is incorporated.

(7) The proportionality constant between the TKN concentration associated with stored COD and the stored COD concentration, f_{ns}, is reformulated as discussed in Section 5.6 (Eq (3.60)).

In order to establish the values of some of the numerical constants, Dold *et al* (1980) carried out experiments using single and series reactor systems under square wave cyclic flow and load conditions. The experiments were carried out at various sludge ages and at temperatures ranging from 12 to 20°C. The constants were determined by trial and error simulations of measured variables under cyclic flow and load conditions until a minimum difference between observations and predictions was obtained.

A complete list of all the kinetic constants for the aerobic activated sludge process as well as constants relative to sewage and sludge characteristics is given in Table 3.7

10.3 Unsteady state solution

10.3.1 Transient or unsteady state conditions

In order to determine a particular kinetic constant of the activate sludge process, it is often useful to run the plant until "steady state" is shown conclusively to have been established and then to impose a quantitative operational step change or an impulse loading condition on the plant to create an unsteady or transient state. Before discussing the solution procedure for activated sludge kinetics under unsteady state conditions it is of value to review briefly the reasons why the creation of an unsteady state in the process is so useful for the determination of kinetic constants.

Under constant flow and load conditions the utilization of the substrates (easily and slowly biodegradable organic material and ammonia) usually appears to be either virtually complete or not to take place at all. If the utilization of a substrate is nearly complete it is not possible to establish the actual value of the corresponding utilization rate constant with any reasonable degree of accuracy. A pertinent example which illustrates this is nitrification; once it occurs is virtually complete and a range of values for the kinetic constant (μ_{nmT}) will give rise to apparently valid solutions. Indeed the only possible way to evaluate the value of μ_{nmT} under steady state conditions is to operate the system near the minimum sludge age for nitrification, but it has been found repeatedly that nitrification then becomes unstable to such a degree that a precise evaluation of μ_{nmT} remains a problem.

Even for heterotrophic sludge mass kinetics steady state operation has been useful only for determining some of the influent sewage characteristics (f_{us}, f_{up}) and a sludge mass parameter (P). The restricted value of the steady state conditions in kinetic studies is illustrated

Table 3.7 Values of mass parameters and kinetic constants and their temperature dependencies **

Carbonaceous material degradation kinetics

Symbol	Value	Unit	Temp.dependency
K_{v20} *	0,25	ℓ.mg VSS^{-1}.d^{-1}	1,10
K_{ms20} *	8,0	mg VSS.mg COD^{-1}	1,20
K_{ss20} *	5,0	mg COD.ℓ^{-1}	1,00
K_{mp20} *	3,0	mg VSS.mg COD^{-1}	1,06
K_{sp20} *	0,04	mg COD.mg VSS^{-1}	0,90
b'_{hT20}	0,62	mg VSS.mg VSS^{-1}.d^{-1}	1,029
P	1,48	mg COD.mg VSS^{-1}	1,00
Y_h	0,45	mg VSS.mg COD^{-1}	1,00
f'	0,08	mg VSS.mg VSS^{-1}	1,00
F_{ma}	1,0	mg VSS.mg VSS^{-1}	1,00

Nitrogenous material degradation kinetics

Symbol	Value	Unit	Temp.dependency
μ_{nm} *	0,33-0,65	mg X_n.mg X_n^{-1}.d^{-1}	1,123
K_{n20} *	1,0	mg NH_3-N.ℓ^{-1}	1,123
b_{n20} *	0,04	mg X_n.mg X_n.d^{-1}	1,029
K_{f20} *	0,023	ℓ.mg VSS^{-1}.d^{-1}	1,10
Y_n	0,1	mg VSS.mg NH_3-N^{-1}	1,00
f_n	0,1	mg NH_3-N.mg VSS^{-1}	1,00

* = values determined by simulation

** = temperature range 12 to 20°C

The value of f_{ns} is calculated from Eq. (3.59a) Section 5

in the evaluation of the yield coefficients for heterotrophic, Y_h, and the endogenous respiration rate constant, b_{hT}: Due to a compensating interdependence of these two constants, sets of associated pairs of values for Y_h and b_{hT} can be found that give virtually the same solution for the steady state process (for example b_{h2o} = 0,10 d^{-1} and Y_h = 0,32 mg VSS.mg COD^{-1} and b_{h20}=0,24 d^{-1} and Y_h = 0,45 mg VSS.mg COD^{-1}. In order to positively identify the appropriate pair a transient state had to be imposed by interrupting the feed and carrying out batch aerobic digestion tests on the mixed liquor from which b_{hT} was identified and Y_h accordingly evaluated (Marais and Ekama, 1976).

In practical terms to evaluate the values of the kinetic constants for utilization of substrates (K_{msT}, K_{ssT}, K_{mpT}, K_{spT}, F_{ma}, μ_{nmT}, K_{nT}), it was found necessary to impose daily cyclic load and flow conditions until dynamic steady state was obtained. By trial and error simulation, curve fitting and critical examination of the effect of the different constants on the response inclusion of the constants in the model could be justified and their magnitude estimated. This approach is quite general: For example the bisubstrate theory developed from the need to explain the oxygen uptake rate profile under square wave loading conditions and the value of the constant (in this case the fraction of easily biodegradable material, f_{ca}) was determined by the value that gave the best correlation between experimental and simulated oxygen uptake rate data.

In obtaining the values of the kinetic constants under dynamic steady state, a new problem was that often a range of values could adequately describe the behavioural response so that even dynamic steady state conditions sometimes did not lead to conclusive estimates of the values for the constants. An example here is the determination of the substrate utilization rate constant for easily biodegradable material, K_{msT}. The rate of utilization of easily biodegradable material is so high that its concentration in completely mixed reactors is negligible irrespective of the loading pattern. This behaviour can be simulated with any K_{msT} value greater than a certain minimum value and consequently no accurate numerical value for K_{msT} can be established by imposing cyclic flow and load conditions. Furthermore severe changes in cyclic flow and load conditions tend to result in hydraulic effects that mask the

rates of changes of concentrations mediated by the biological reactions. The alternative is to keep the input flow and load conditions constant and to impose spatial concentration gradients by subdivision of the process volume into a series of reactors. This is not as successful as one would expect because the hydraulic regime tends to approach that of a plug flow regime, a regime that is very sensitive to small perturbations in flow or influent concentrations or fluctuations in the kinetic constants. Consequently an unstable experimental response is often observed. Hence in practice processes with a very severe cyclic variation of flow and/or load and with a large number of reactors does not necessarily provide a means for a precise evaluation of all the constants.

The ideal solution would be to devise a system that tends to give stable response and to impose a perturbation, measuring the effect or effects of the transient condition created by the perturbation. By appropriately selecting the type and severity of the perturbation it should be possible to isolate a transient condition that principally affects one reaction in the system and thus to isolate and determine the kinetic constant or constants associated with that reaction. For example, by introducing a pulse of ammonia into an aerobic reactor under constant flow and load conditions, the nitrification rate is increased and the kinetic constant μ_{nmT} may be evaluated by observing one or more of the variables that are affected by nitrification (oxygen uptake rate, ammonia concentration, nitrate concentration, alkalinity).

Care must be taken that the step change imposed on the system does not result in a qualitative change of the behavioural pattern of the organisms. For example, if μ_{nmT} is to be determined by adding an ammonia pulse to a reactor, the introduced ammonia mass must not be so large that it affects the pH of the mixed liquor or becomes toxic for the live cells. Usually it is possible to create a transient state that allows the precise determination of kinetic constants to be made without the need of a step change so great that it might affect the behavioural pattern of the process as a whole.

Different possibilities of imposing step changes on an activated sludge process exist; basically four different methods can be distinguished: (1) Introduction of a pulse of a particular substrate, (2) Interruption of the influent flow or load during a certain period, (3) Interruption of the air supply to an aerobic reactor for a certain period, (4) Imposing a transient change on the sludge age at some point in time by bypassing the settler, i.e. transforming the system to a flow through system, keeping the feed rate constant. The selection of the method of imposing the perturbation will depend upon the kinetic constant that needs to be determined. In Chapter 4 it will be shown that for the determination of the most important kinetic constants in nitrification-denitrification processes the interruption of the air supply and of the influent flow supply are particularly useful.

A problem of a different nature that the modeller must take cognisance of continuously, is that the basic model may be deficient, for if this is true, evidently, no consistency can be obtained between simulation and observation. The reactions occurring in an activated sludge process are so complex that an exact mathematical description is unlikely, so that hypotheses about the behaviour of the organism mass must be made. The reasonableness of such hypotheses can be assessed only by comparing simulated and experimental response over a wide range of operational conditions (sludge age, temperature influent flow and load pattern) and sewage characteristics (settled or raw sewage). If no consistency can be obtained (between simulation and observation), efforts should be directed towards improving the model rather than attempting to obtain improved values for the constants. Once reasonable certitude as to the basic structure of the model has been obtained, methods for improving the estimates of the constants become significant.

10.3.1 Solution procedure for unsteady state

The solution procedure for the process under unsteady or transient conditions differs from that for the process under dynamic steady state conditions (Section 10.2) in that the starting values of the variables for the transient process are either measured (all the measurable variables) or estimated on the basis of steady state or dynamic steady state prevailing

before the transient condition is imposed.

Starting values of some variables may be accepted without further consideration. These are:

(1) Easily biodegradable material.

(2) Unbiodegradable soluble nitrogen: Almost invariably the concentrations of these variables are so small that these can be equated to zero. Starting values that can be obtained from direct measurement are:

(1) Unbiodegradable soluble COD, S_u (the filtered COD concentration in a reactor or in the effluent).

(2) Ammonia concentration, N_a.

(3) Organic nitrogen concentration, N_o. (The difference between the filtered TKN and the ammonia concentration).

(4) Nitrate concentration, N_n,

(5) Alkalinity, Alk.

The values of the variables that cannot be measured (S_{bp}, X_s, X_a, X_e, X_i, X_n) may be obtained by theoretical calculations using the expressions developed in Sections 9.3 and 9.4 in the case of a single reactor process or by simulation using the dynamic steady state program in the case of a multi reactor process. Once the starting values for the different variables in the reactors of the series system have been established simulations are carried out and the kinetic constant or constants are determined by fitting simulated data with observed data that are obtained during the period following the perturbation that is imposed on the system. The solution procedure for the unsteady state has some distinct advantages over the procedure for dynamic steady state, discussed in Section 10.2:

(1) The unsteady state solution procedure is not restricted to cyclic flow and load patterns: Calculations of the response of true unsteady state behaviour is possible.

(2) The starting values of all the measurable variables are the true values as measured in the system.

(3) The magnitide of imposed transient usually can be selected

to give a precise value for the required kinetic constant.

(4) Only one iteration of calculations is carried out over the periods that measurements are done, thus reducing considerably the required CPU time.

The listing of the program for unsteady state is in Appendix A2.

CHAPTER FOUR

GENERAL THEORY FOR DENITRIFICATION

1. THEORETICAL INTERPRETATION OF DENITRIFICATION BEHAVIOUR

In Chapter 2 an empirical model was presented to describe the rate and extent of denitrification in single sludge nitrification-denitrification systems with pre- and/or post-denitrification reactors under constant flow and load conditions. It was shown that under the limitation of constant flow and load conditions, the empirical model gives a good account of denitrification behaviour and allows quantitative incorporation of the effects of sludge age, temperature, influent COD and position of the anoxic reactors. However under cyclic flow and load conditions the model does not describe denitrification behaviour correctly. Under cyclic flow and load conditions the rate of nitrate removal is high during periods of peak flow or load, and low during periods of low flow or load. In order to explain quantitatively such variations in the rate of denitrification, work was commenced to construct a general denitrification model.

Because denitrification is mediated by heterotrophic organisms with nitrate or nitrite replacing oxygen as an electron acceptor, it was hypothesized that the behaviour of the organisms under denitrification (anoxic) conditions should, qualitatively at least, not differ from that under aerobic conditions, although quantitatively the behaviour might show differences. If this hypothesis is valid, then one could expect that dissolved oxygen (DO) consumption for degradation of organic material in aerobic plug flow reactors would exhibit a behaviour similar to that of nitrate removal in anoxic plug flow reactors if both were operated under the same constant flow and load conditions i.e. one would expect a two phase DO-time profile in a "pre-aerobic" plug flow reactor similar to the two phase NO_3-time profile in a predenitrification plug flow reactor (Fig 2.16) and a single

DO-time profile in a "post-aerobic" plug flow reactor akin to the single phase NO_3-time profile in a postdenitrification reactor (Fig 2.17).

Accordingly a computer program was written for aerobic plug flow reactors using the basic differential equations for the reaction rates (Eqs 3.86 to 3.95) and simulations of DO-time profiles in pre- and post-aerobic plug flow reactors under constant flow and load conditions were carried out. The simulated response of the dissolved oxygen profile in the pre- and post-aerobic plug flow reactors for X_a = 1000 mg VSS.ℓ^{-1}, sludge age = 20d, temperature = 20°C is shown in Figs 4.1a and b respectively. (The μ_{nmT} value in the simulations was kept at zero so that no nitrification took place and the oxygen consumption was due solely to degradation of organic material). Also shown in Figs 4.1 (a and b) are the nitrate profiles based on the average experimental values of the empirical denitrification rate constants and plotted as equivalent oxygen*, to allow direct comparison between the respective curves. The simulated aerobic and observed anoxic profiles are strikingly similar: the two phase behaviour in the pre-anoxic reactor and the single phase behaviour in the post-anoxic reactor are clearly reproduced in the aerobic simulations. Also the concentration of nitrate removed (as equivalent oxygen) due to the high rate reaction in the primary phase of a predenitrification reactor is the same as the corresponding concentration of oxygen removed in the aerobic reactor (see extrapolation to t=0 in Fig 4.1a). The similarity of the profiles was so clearly evident that it was accepted that the general aerobic model, described in Chapter 3, could provide a basis for modelling denitrification behaviour.

However from Figs 4.1 (a and b) it was apparent that there was a quantitative difference between aerobic and anoxic reactors. The observed rate of nitrate removal (as equivalent oxygen) in an anoxic reactor was lower than the rate of oxygen consumption in a comparable aerobic reactor. The problem was now: what changes are required to adapt the differential equation describing the aerobic activated sludge process in order to make these applicable in an anoxic environment?

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*Adjustment on the basis of 1 mg NO_3-N ≡ 2,86 mg O_2

Accepting that the structure of the kinetic relationships in an anoxic environment remained the same as in an aerobic environment, the computer program describing the aerobic activated sludge process was rewritten for anoxic reactors, with the following modifications :

(1) The oxygen uptake rate was converted into an equivalent oxygen uptake rate or a nitrate removal rate, i.e.

$$O_{c,eq} = -2,86\,(dN_n/dt)_r = (1-PY_h)(r_{us}+r_{up}) \qquad (4.1)$$

where

$(dN_n/dt)_r$ = denitrification rate

$O_{c,eq}$ = equivalent oxygen uptake rate.

Hence

$$r_{den} \triangleq (dN_n/dt)_r = -(1-PY_h)(r_{us}+r_{up})/2,86 \qquad (4.2)$$

One constraint that must be imposed is that Eq (4.2) applies only if nitrate is present in the anoxic reactor i.e. $N_n > 0$. When $N_n = 0$ no utilization of substrate can take place and $r_{us}=r_{up}=0$.

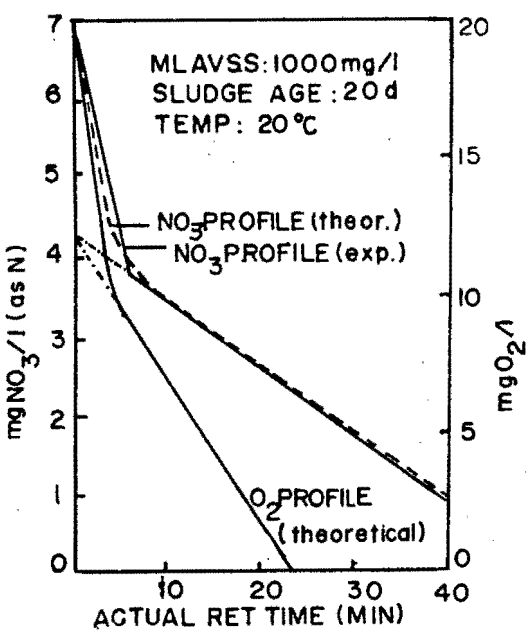

Fig. 4.1a *Predenitrification.*

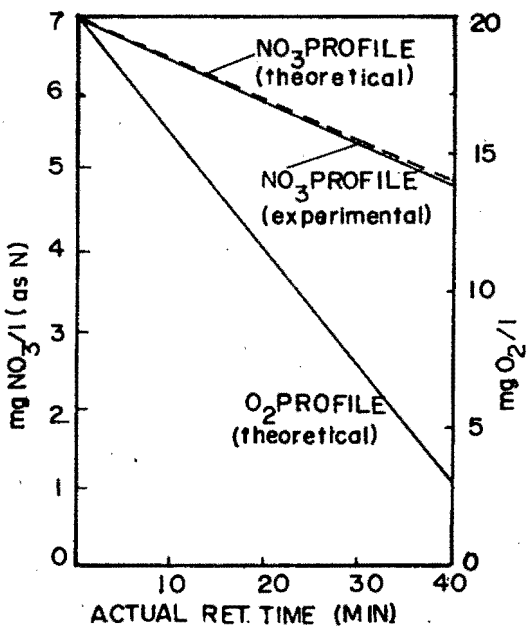

Fig. 4.1b *Post-denitrification.*

Fig. 4.1 *Experimental and simulated NO_3 - time profiles in an anoxic plug flow reactor and DO profile in an aerobic plug flow reactor.*

(2) Nitrification does not take place in an anoxic environment i.e. $(\mu_{nm})_{anoxic} = 0$.

(3) The alkalinity increases due to denitrification. From stoichiometric considerations (see Eq 2.26) there is an increase of 3,57mg $CaCO_3$ alkalinity per mg NO_3-N denitrified. In aerobic systems the rate of change of alkalinity depends upon the rates of nitrification and of ammonification (Eq 3.95).

$$(r_{alk})_{aerobic} = -7,14\ r_n + 3,57\ [r_{co} - f_n Y_h (r_{us} + r_{up})] \qquad (3.95)$$

In an anoxic environment no nitrification takes place i.e. $r_n = 0$ but ammonification continues. Also there is a rate of increase of alkalinity due to denitrification equal to 3,57 times the rate of denitrification, r_{den}. Hence combining Eqs 4.2 and 3.95:

$$(r_{alk})_{anoxic} = 3,57\ [r_{co} - f_n Y_h (r_{us} + r_{up})] + 3,57 (r_{us} + r_{up})(1 - PY_h)/2,86$$

$$= 3,57\ [r_{co} - (r_{us} + r_{up})(f_n Y_h - (1 - PY_h)/2,86)] \qquad (4.3)$$

When these changes were incorporated in the general model and the nitrate removal response in anoxic plug flow reactors was simulated, the simulation reproduced qualitatively the denitrification behaviour. However, the observed rate of nitrate removal in the secondary phase of a predenitrification reactor and in a post-denitrification reactor was not as rapid as the simulated decrease of nitrate concentration. This indicated that the theoretical rates of utilization of organic material in the anoxic reactors were too high; see Eq(4.2.) To determine the rate constants under anoxic conditions, it was considered that the degree to which the rate of utilization of slowly biodegradable material should be reduced would be best determined by analysing the behaviour of the post-denitrification reactor. In a post-denitrification reactor the concentration of easily biodegradable material is virtually zero and the rate of nitrate removal is associated almost entirely with the utilization of slowly biodegradable material. It was found possible to simulate accurately the experimental nitrate profile in a post-denitrification reactor at 20°C if the value of the specific substrate utilization rate constant for slowly biodegradable material

was reduced to a fraction η = 0,38 of the value this constant has in an aerobic environment. This is illustrated in Fig 4.1b which shows the simulated nitrate profile in a post-denitrification plug flow reactor T = 20°C, R_s = 20d, X_a = 1000 mg VSS.ℓ^{-1} and an anoxic sludge fraction of 25 per cent, when the specific substrate utilization rate constant for slowly biodegradable material in an anoxic environment, K^1_{mp}, is set at a value

$$K^1_{mp} = \eta.K_{mp} = 1,14 \text{ mg COD.mg } X_a^{-1}d^{-1}(20^oC) \qquad (4.4)$$

where η = 0,38. When this value for K^1_{mp} was used to simulate denitrification in a predenitrification plug flow reactor under the same conditions of temperature, sludge age and anoxic volume fraction, again a very close correspondence between simulated and average experimental nitrate profiles in the secondary phase was obtained (see Fig 4.1a).

With regard to the specific utilization rate constant for easily biodegradable material, it had not been possible to determine K_{ms} accurately in an aerobic environment from the completely mixed reactor response under square wave cyclic flow and load conditions (see Section 10.3.1 Chapter 3); the utilization of easily biodegradable material was found to be an extremely rapid reaction and all that could be established was a lower limit for the constant - any value greater than the minimum predicted a behaviour in conformity with the observation. The lower limit was determined as K_{ms} = 5 mg COD.mg $X_a^{-1}.d^{-1}$ at 20°C. However, in anoxic plug flow reactors it was possible to derive a value for K_{ms} from fitting simulated and experimental nitrate profiles in the plug flow reactor. This gives K_{ms} = 8 mg COD.mg $X_a^{-1}.d^{-1}$ at 20°C. Consequently, this value, because it also satisfies the observed aerobic response, was adopted for both aerobic and anoxic conditions. Uncertainty in the value of K_{ms} in both aerobic or anoxic states, in fact, is not important, because the rate of utilization of easily biodegradable material is so high that the reaction is complete well within the actual retention times normally provided (i.e. $R_1 >> R_{min}$).

Once the values of K_{ms} and K^1_{mp} at 20°C were established for a sludge age = 20 d and an anoxic volume fraction = 25 per cent,

simulations were repeated for different sludge ages and anoxic sludge fractions of plug flow denitrification reactors. From the simulated slopes of the nitrate profiles in the secondary phase of the pre-denitrification reactor and in the post-denitrification reactor, the values of K_2 and K_3 respectively were calculated. The simulated response data indicated that (1) the magnitudes of the anoxic volume fractions have virtually no effect on the simulated K values, neither in the pre- nor in the post-denitrification reactors. (2) In the range of sludge ages from 10 to 20 days the respective K values remain virtually constant (see Fig 4.2a and b). The constancy of the K values over this range of sludge ages had previously been noted experimentally by Marais and his co-workers and was also apparent in the analysis of the work by Sutton *et al* (1979) (Fig 2.21).

To establish the temperature dependency for the constant K^1_{mp}, the simulations were repeated for a temperature of 14°C using pre- and post-denitrification plug flow reactors. The value of K^1_{mp} was varied until close correspondence between experimental and observed nitrate profiles at that temperature was obtained. As for the aerobic model, an Arrhenius type of temperature dependency was assumed, i.e.

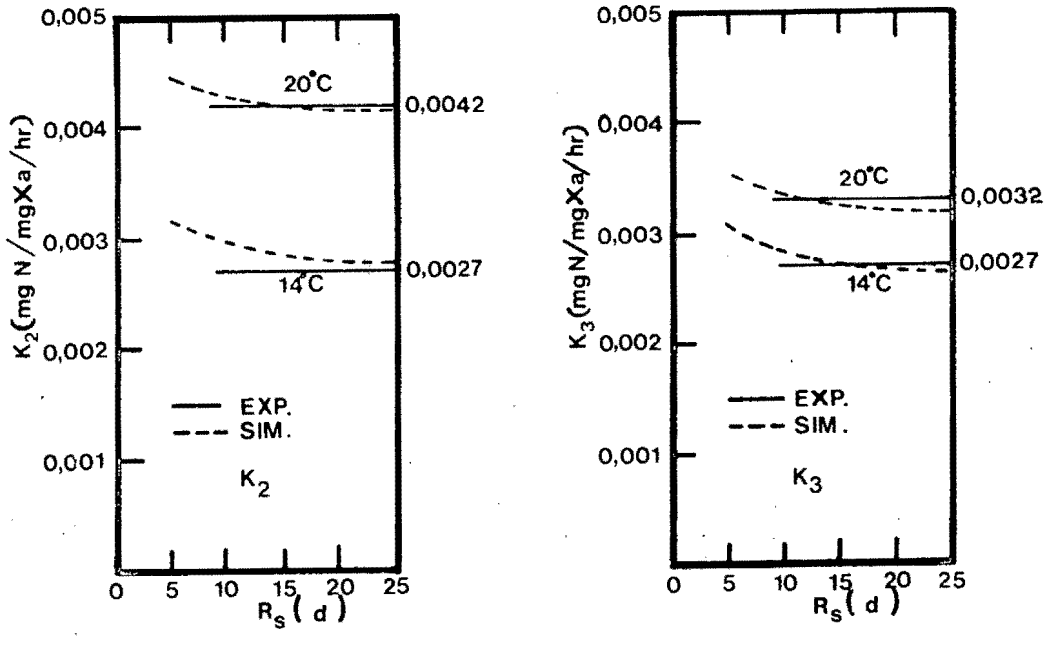

Fig. 4.2 *Simulated and experimental denitrification rate constants K_2(Fig. 4.2a left hand side) and K_3(Fig. 4.2b right hand side) versus sludge age.*

$$K^{1}_{mpT} = K^{1}_{mp20}\ \theta_p^{T-20} \qquad (4.5)$$

It was found that at 14°C the best correspondence between the experimental and simulated K_2 and K_3 values was obtained when K^{1}_{mpT} was given a temperature dependency of $\theta_p = 1{,}06$ both for the pre- and post-denitrification reactors. In Figs 4.2 (a and b) experimental and simulated values of K_2 and K_3 for a temperature of 14°C are shown plotted as a function of the sludge age. For $\theta_p = 1{,}06$ there is good correlation of the observed and theoretical K values over the range T = 14° to 20°C. Dold *et al* (1980) accepted a temperature dependency of $\theta_p = 1{,}029$ for K_{mp} in their analysis of aerobic systems. However, when the temperature dependency $\theta_p = 1{,}06$ was inserted for K_{mp} to simulate experimental data of aerobic systems at 12°C, it was found that the predicted response was fractionally closer to the observed one than in the case where $\theta_p = 1{,}029$ was used. On the basis of this observation the value of $\theta_p = 1{,}06$ seems to be acceptable for both aerobic and anoxic environments.

The temperature dependency of K_{msT} was also determined by fitting the simulated to the observed profiles in the primary phase of a plug flow predenitrification reactor at different temperatures. This indicated a temperature dependency for K_{msT} of:

$$K_{msT} = K_{ms20}(\theta)_s^{T-20}$$

$$= K_{ms20}(1{,}20)^{T-20} \qquad (4.6)$$

Once the temperature dependencies of K^{1}_{mp} and K_{msT} were estimated, simulations of plug flow denitrification were done over a temperature range between 12 and 22°C and the simulated denitrification rate constants K_1, K_2 and K_3 were calculated from the slopes of the generated nitrate-retention time profiles. In Fig 4.3 the simulated values of K_1, K_2 and K_3 are shown plotted as a function of the temperature for sludge ages R_s = 10 d and R_s = 20 d over a temperature range from 12°C to 22°C. Experimentally observed denitrification rate constants are also indicated.

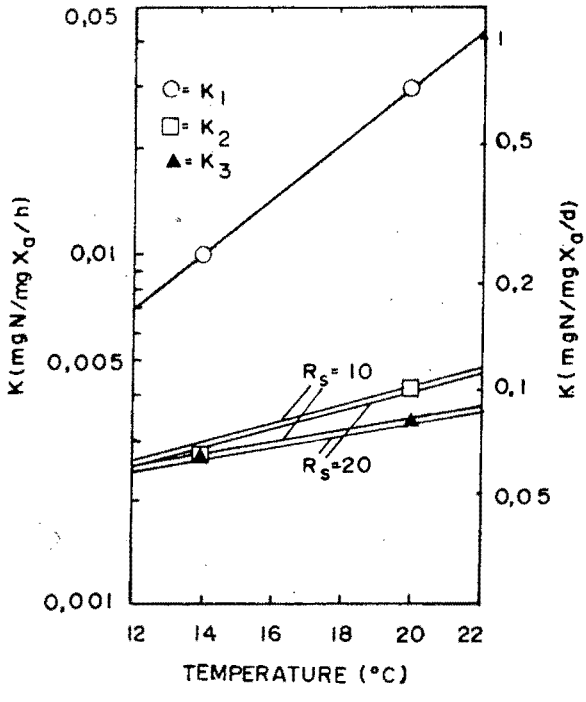

Fig. 4.3 *Simulated and experimental denitrification rate constants K_1, K_2 and K_3 versus temperature for R_s = 10 d and R_s = 20 d.*

The procedure described above, to adapt the general aerobic model to describe denitrification behaviour, and, the calibration of this model against experimentally observed data, allowed the formulation of a general kinetic model for the kinetics of the activated sludge process in an anoxic environment, capable of *inter alia* simulating qualitatively and quantitatively the denitrification behaviour, observed in all the experimental work discussed in Chapter 2.

These results indicated that formulations of a general model for an anoxic environment could be achieved by using the same basic kinetic relationships describing the activated sludge process in an aerobic environment, provided suitable adaptations were made. These adaptations were not unexpected, being due either to the nature of the anoxic environment,or a quantitative difference in response in an anoxic and in an aerobic environment. The adaptations due to the nature of the anoxic environment included: (1) nitrate is the final electron acceptor, (2) no nitrification takes place and (3) the

alkalinity increases due to denitrification. The adaptation due to the quantitative difference in the response of anoxic and aerobic processes respectively was that the value of the substrate utilization rate constant for slowly biodegradable material, K_{mp20}, needed to be reduced from 3,0 in an aerobic environment to a value of 1,14 mg COD. mg $X_a^{-1}.d^{-1}$ in an anoxic environment (designated in the following as K^1_{mp}) i.e. $K^1_{mp} = 0,38\ K_{mp}$.

In Table 4.1 the kinetic expressions are presented describing the activated sludge process in an anoxic environment. The corresponding numerical values of the mass parameters and kinetic constants are listed in Table 4.2.

Having thus formulated a general model to describe denitrification it is now of interest to enquire (1) if the empirical denitrification rate constants can be linked to the basic kinetic constants of the activated sludge process and (2) if the denitrification behaviour under general conditions of flow and load can be described by the present model. These points will be discussed in Sections 2 and 3 respectively.

2. THEORETICAL INTERPRETATION OF THE EMPIRICAL DENITRIFICATION RATE CONSTANTS

Having calibrated the model it is now of interest to inquire if there is a quantitative link between the empirical K values (K_1, K_2 and K_3) and the kinetic constants for substrate utilization. This can be done as follows: The rate of nitrate utilization as expressed in Eq (4.21) can be divided into a rate associated with the utilization of easily biodegradable material, dN_1/dt, and a rate associated with the utilization of slowly biodegradable material, dN_2/dt, where

$$dN_1/dt = -(1-PY_h)r_{us}/2,86 \qquad (4.37)$$

$$dN_2/dt = -(1-PY_h)r_{up}/2,86 \qquad (4.38)$$

i.e.

Table 4.1 Differential equations describing reaction rates in the anoxic activated sludge process

$$r_{us} = [K_{msT} \cdot S_{bs} \cdot / (K_{ssT} + S_{bs})] X_a \quad (4.7)$$

$$r_a = [K_{vT} \cdot S_{bp} \cdot (F_{ma} - X_s / X_a)] X_a \quad (4.8)$$

$$r_{up} = [K^1_{mpT} \, X_s P / (X_s P + K_{spT} X_a)] X_a \quad (4.9)$$

$$r_d = [\, b'_{bT}] X_a \quad (4.10)$$

$$r_{dn} = [\, b_{nT}] X_n \quad (4.11)$$

$$r_{co} = [\, K_{rt} N_o] X_a \quad (4.12)$$

$$r_{sbs} = - r_{us} \quad (4.13)$$

$$r_{sbp} = - r_a + P(1-f') b_{hT} r_d \quad (4.14)$$

$$r_{xs} = (r_a - r_{up}) / P \quad (4.15)$$

$$r_{xa} = Y_h (r_{us} + r_{up}) - r_d \quad (4.16)$$

$$r_{xe} = f' r_d \quad (4.17)$$

$$r_{xn} = - r_{dn} \quad (4.18)$$

$$r_{na} = - f_n Y_h (r_{us} + r_{up}) + r_{co} \quad (4.19)$$

$$r_{no} = - r_{co} + f_n (1-f\,) r_d + f_{ns} \, (r_{up} - r_a) \quad (4.20)$$

$$r_{den} = -(1-PY_h)(r_{us} + r_{up}) / 2{,}86 \quad (4.21)$$

$$r_{alk} = 3{,}57 \{ r_{co} - (r_{us} + r_{up}) [f_n Y_h - (1-pY_h) / 2{,}86] \} \quad (4.22)$$

Table 4.2 Mass parameters and kinetic constants in the anoxic activated sludge process

Mass parameters

Y_h = 0,45 mg X_a.mg COD^{-1} (4.23)

P = 1,48 mg VSS.mg COD^{-1} (4.24)

f' = 0,08 mg VSS.mg X_a^{-1} (4.25)

f_n = 0,10 mg N.mg VSS^{-1} (4.26)

Y_n = 0,04 mg X_n.mg N^{-1} (4.27)

F_{ma} = 1,0 mg VSS.mg VSS^{-1} (4.28)

Kinetic constants

$K_{msT} = 8(1,20)^{T-20}$ mg COD.mg $X_a^{-1}.d^{-1}$ (4.29)

$K^1_{mpT} = 1,14(1,10)^{T-20}$ mg COD.mg $X_a^{-1}.d^{-1}$ (4.30)

$K_{ssT} = 5$ mg COD.ℓ^{-1} (4.31)

$K_{spT} = 0,04(0,90)^{T-20}$ mg COD.mg X_a^{-1} (4.32)

$K_{vT} = 0,25(1,10)^{T-20}$ ℓ.mg $X_a^{-1}.d^{-1}$ (4.33)

$b'_{hT} = 0,62(1,029)^{T-20}$ mg X_a.mg $X_a^{-1}.d^{-1}$ (4.34)

$K_{rT} = 0,023(1,10)^{T-20}$ ℓ.mg $X_a^{-1}.d^{-1}$ (4.35)

$b_{nT} = 0,04(1,029)^{T-20}$ mg X_n.mg $X_n^{-1}.d^{-1}$ (4.36)

$$r_{den} = dN_1/dt + dN_2/dt \quad (4.39)$$

(1) Analysis of (dN_1/dt) :

Substituting for r_{us} from Eq (4.7), one obtains the following expression for (dN_1/dt) :

$$dN_1/dt = -[(1-PY_h)\ K_{msT}\ S_{bs}\ X_a/((S_{bs}+K_{ssT})\ 2,86)] \quad (4.40)$$

As the half saturation value, K_{ssT}, is very small, (K_{ssT} = 5 mg COD.ℓ^{-1}) the expression $K_{ms}.S_{bs}/(K_{ssT}+S_{bs})$ remains virtually constant and equal to K_{ms} until S_{bs} falls to very low values. Consequently, the function in square brackets in Eq (4.40) remains virtually constant and hence

$$dN_1/dt = \Delta N'_{1a}/t_p = -K_1 X_a \quad (4.41)$$

Where

$\Delta N'_{1a}/t_p$ = reactor nitrate removal due to utilization of easily biodegradable material

t_p = duration of the primary phase

$$K_1 = (1-PY_h).K_{msT}.S_{bs}/(K_{ssT}+S_{bs})/2,86 \quad (4.42)$$

As

$$S_{bs}/(K_{ssT}+S_{bs}) \simeq 1 \quad (4.43)$$

hence,

$$K_1 = (1-PY_h).K_{msT}/2,86 \quad (4.44)$$

From Eq (4.2) the *mass* of denitrified nitrate is directly proportional to the *mass* of substrate utilized in the denitrification reaction, the proportionality constant being $(1-PY_h)/2,86$. In a pre-denitrification reactor normally all the influent easily biodegradable substrate is completely utilized and hence the nitrate removal associated with the utilization of easily biodegradable substrate can be expressed as:

$$\Delta N'_{ls} = (1-PY_h).S_{bsi}/2,86 \quad (4.45)$$

where

$\Delta N'_{ls}$ = system nitrate removal due to utilization of easily biodegradable material

S_{bsi} = influent easily biodegradable material concentration.

Substituting for Y_h = 0,45 mg VSS. mg COD^{-1} and P = 1,48 mg COD. mg VSS^{-1}:

$$\Delta N'_{ls} = 0,117\ S_{bsi} \quad (4.46)$$

Dold *et al* (1980), from observations of oxygen uptake rate profiles in aerobic processes under square wave loading conditions, concluded that the concentration of easily biodegradable influent COD in domestic waste water is approximately 24 per cent of the total biodegradable influent COD:

$$S_{bsi} = f_{ca}.S_{bi} \quad (4.47)$$

where

f_{ca} = ratio of easily biodegradable to total biodegradable influent COD concentration

= 0,24 mg COD. mg COD^{-1}.

Substituting Eq (4.47) in Eq (4.46):

$$\Delta N'_{ls} = 0,028.S_{bi}. \quad (4.48)$$

Hence from theoretical considerations and accepting the value of f_{ca} reported by Dold *et al* (1980), there is a proportional relationship between the system nitrate removal in the primary phase, $\Delta N'_{ls}$, and the influent biodegradable COD concentration. This proportional relationship has also been observed experimentally and the

experimentally found proportionality constant is identical to the theoretical value calculated above (see Fig 2.20).

(2) Analysis of (dN_2/dt) :

With regard to the slowly biodegradable COD fraction in the predenitrification reactor, the nitrate utilization rate can be expressed by substituting Eq (4.9) in Eq (4.38)

$$dN_2/dt = -(1-PY_h)K^1_{mpT}.P.X_s.X_a/(PX_s+K_{spT}X_a)/2,86 \qquad (4.49)$$

In the predenitrification reactor, X_s is large and due to the slow rate of the reaction there is only a small change in X_s between the beginning and the end of the plug flow reactor. Consequently, again it is possible to write as a good approximation :

$$dN_2/dt = -\Delta N''_{1a}/R_{1a} = -K_2X_a \qquad (4.50)$$

Considering the post-denitrification reactor, X_s generally is very small, arising principally from storage of lysis products. The concentration of X_s hardly changes from the beginning to the end of the reactor and consequently again the slope, dN_2/dt, can be expressed as in Eq (4.50) :

$$dN_3/dt = \Delta N_{3a}/R_{3a} = -K_3X_a \qquad (4.51)$$

From the above it is clear that the empirical denitrification rate constants K_1, K_2 and K_3 as well as the empirical proportionality constant α can be expressed in terms of the basic equations for rates of utilization of easily and slowly biodegradable material and the fraction of easily biodegradable influent material, f_{ca}. This conclusion provides a strong indication that the hypothesis required to integrate denitrification kinetics into the general theory of activated sludge, namely that the two phase nitrate profile in a predenitrification reactor is due to the bisubstrate nature of domestic

sewage, is indeed justified.

The behavioural patterns described above have been observed elsewhere: Heideman (1979) analysed the denitrification behaviour in batch experiments, using mixtures of sludge and influent municipal sewage. By observing the change of nitrate and soluble COD with time he found that (1) the nitrate time profile was very similar to that observed by Marais *et al* in predenitrification plug flow reactors, i.e. a high denitrification rate in the primary phase followed by a lower rate in the secondary phase. (2) Cessation of the primary phase occurred when the soluble COD concentration attained a constant value, i.e. when all the easily biodegradable substrate was utilized. (3) The ratio between the nitrate removal due to the high rate, i.e. $\Delta N'_{1s}$ and the decrease in the soluble COD concentration, i.e. S_{bsi}, was repeatedly found to be in the range 0,11 to 0,12 mg N. mg COD^{-1}, virtually identical to the value determined above (Eq 4.46). (4) The two phase behaviour was observed only when sludge and influent sewage were mixed; batch denitrification tests using an activated sludge sample only yielded linear profiles akin to the post-denitrification plug flow nitrate profiles. Unfortunately from the data supplied by Heideman for the batch tests, it is not possible to calculate the active sludge concentration so that the denitrification rate constants (in terms of mg NO_3-N. mg $X_a^{-1}.d^{-1}$) cannot be determined.

The analysis above indicates that the denitrification "constants" K_1, K_2 and K_3 have no fundamental kinetic significance; the constants are a result of a combination of reactions which, fortuitously, show little variation with sludge age (in the range from 10 to 20 days) at any selected temperature in the range from 12°C to 22°C. Despite this,both from an experimental and a practical point of view, it would appear that for design the acceptance of constant values for K_1, K_2 and K_3 at any selected temperature is not unreasonable and is an allowable approximation for estimating the denitrification achievable under constant flow and load conditions.

3. MODEL VERIFICATION

In Section 1 it was shown how the general model for

nitrification-denitrification kinetics was calibrated against experimental results obtained on anoxic plug flow reactors under constant flow and load conditions. It now remains to check if the calibrated model simulates the behaviour of completely mixed series reactor systems under more general patterns of flow and load.

The predictive power of the calibrated model was tested by comparing simulated data with those observed in completely mixed series reactor nitrification-denitrification systems under a variety of flow and load conditions (space dependent-time independent and space dependent-time dependent systems), and in a single reactor system where the environment was made sequentially aerobic and anoxic under constant flow and load conditions.

3.1 Space dependent-time independent systems

A space dependent-time independent system can be established by having a series of reactors receiving a constant influent flow and load. Experiments using predenitrification systems under constant flow and load conditions were carried out by Ekama and Marais (1976), both at bench and at pilot scale.

The bench scale system was a predenitrification system composed of three completely mixed reactors: the first (2,5ℓ) anoxic, the second (2,5ℓ) aerobic and the third (7,5ℓ) aerobic (Fig 4.4). Two series of tests under constant flow and load conditions were carried out, each series being composed of 13 daily tests. The operational parameters and sewage characteristics that prevailed during the test periods are shown in Table 4.3.

The pilot plant was also a predenitrification system consisting of five equal sized reactors each with a 5 m^3 capacity, the first anoxic and the remainder aerobic (Fig 4.6). With this system two series of tests were also carried out under constant flow and load conditions. In Table 4.4 the operational parameters and sewage characteristics prevailing during the test periods are shown.

Comparing Tables 4.3 and 4.4, it can be noted that the pilot plant was 2 000 times bigger than the bench scale system, but that the operational parameters were the same (hydraulic loading rate,

sludge age, pH) or very similar (organic loading rate, nitrogen loading rate, temperature). The main difference was that the second aerobic reactor in the bench scale unit was equivalent to the last three reactors in the pilot plant system. From the similarity in operational parameters and sewage characteristics, theoretically the behaviour of all the intensive parameters such as concentrations and pH in these two systems should be similar.

The influent for both systems was the same and was obtained as follows: During the daily period of maximum COD load in the waste flow to the sewage works (between 12.00 and 14.00h) sewage was withdrawn from the primary settler of the sewage works and one of two 45 m^3 storage tanks was filled and used as feed the following day, when the other storage tank was filled. The choice of time for filling the tanks (12.00 - 14.00 h) was found by trial and error; during the chosen period there was very little variation from day to day in the values of the influent COD and TKN. The stored volume was sufficient for the feed of both the bench and pilot plant unit over one day. Both systems were fed at constant rate. By this procedure it was possible not only to analyse the nitrification-denitrification behaviour in both systems but also to compare the results of the bench scale systems with the pilot plant system.

Two series of tests were carried out on both the bench scale and the pilot plant system. During each series of tests daily observations were made on the following parameters in both systems :

1. Oxygen uptake rates in the aerobic reactors.
2. COD : influent, effluent and filtered mixed liquor of each reactor.
3. TKN : influent, effluent and filtered mixed liquor of each reactor.
4. NH_3 : effluent and filtered mixed liquor of each reactor.
5. NO_3 : influent, effluent and filtered mixed liquor of each reactor.

6. VSS : concentration in each reactor.

7. Temperature

8. pH

Daily results observed on the bench scale system during the two series of experiments are listed in Appendix A_3. The averages of the results for each of the two series of tests are listed in Tables 4.5a and 4.5b respectively.

Daily results observed on the pilot scale system during the two series of experiments are listed in Appendix A_4. The averages of the results for each of the series of tests are listed in Tables 4.6a and 4.6b respectively. To compare the experimental response with the simulated response using the general model it is necessary first to estimate the sewage characteristics i.e. the fractions of easily and slowly biodegradable and of soluble and particulate unbiodegradable material and the maximum specific growth rate of nitrifiers.

From earlier tests on the same plants under aerobic conditions the ratio of (easily biodegradable material/total biodegradable material) in the influent was estimated at 24 per cent i.e. $f_{ca} = 0,24$.

The fraction (soluble unbiodegradable/total influent COD) i.e. f_{us}, was estimated from the ratio between the effluent and the influent COD : $f_{us} = S_t/S_{ti}$. From Tables (4.5a, 4.5b, 4.6a and 4.6b) for both series of tests on each system the fraction, f_{us}, was $f_{us} = 0,04$ mg COD. mg COD^{-1}.

In order to estimate the fraction of particulate unbiodegradable influent COD, f_{up}, it was assumed that the formulation of Marais and Ekama (1976) and the associated values for the constants Y_h and b_{hT} were applicable to both systems. The value of f_{up} was estimated from the observed sludge production in the systems during the two series of experiments using Eq (3.171 i.e.

$$m_{xv} = (MX_a + MX_e + MX_i)/MS_{ti}$$

$$= Y_h R_s (1+fb_{hT}R_s)(1-f_{us}-P.f_{up})/(1+b_{hT} R_s)+f_{up}R_s. \qquad (4.52)$$

Table 4.3 Operational parameters and average sewage characteristics for the two series of tests using the bench scale predenitrification unit (See Fig 4.4).

Parameters	Series I	Series II
Period of operation	14-10 to 26-10-1976	27-10 to 9-11-76
Configuration	Series pre-denitrification	Series pre-denitrification
Reactor Volume	a) 2,5ℓ (anoxic) b) 2,5ℓ (aerobic) c) 7,5ℓ (aerobic)	a) 2,5ℓ (anoxic) b) 2,5ℓ (aerobic) c) 7,5ℓ (aerobic)
Influent flow ($l.d^{-1}$)	20	20
Influent COD ($mg.\ell^{-1}$)	490	461
Influent TKN ($mgN\ell^{-1}$)	46,7	46,1
Influent pH	7,8	7,8
Temperature (^{o}C)	25,1	23,8
Sludge age (d)	18	18
Recycle ratios		
1) From settler to anoxic reactor	2:1	3:1
2) From first aerobic to anoxic reactor	3:1	3:1

Table 4.4 Operational parameters and sewage characteristics of the pilot plant pre-denitrification unit (See Fig 4.6)

Parameters	Constant load		Cyclic load
	Series I	Series II	
Period of operation	5.10.1976 to 20.10.1976	21.10.1976 to 9.11.1976	
Configuration	Series, prede-nitrification	Series, prede-nitrification	Series, prede-nitrification
Reactor volume	a) 5m³ (anoxic)	a) 5m³ (anoxic)	a) 5m³ (anoxic)
	b) 5m³ (aerobic)	b) 5m³ (aerobic)	b) 5m³ (aerobic)
	c) "	c) "	c) "
	d) "	d) "	d) "
	e) "	e) "	e) "
Influent flow (m³/d)	40	40	60
Influent COD (mg/ℓ)	477	461	mean: 363
Influent TKN (mg/ℓ)	45,1	46,6	mean: 33,5
Influent pH	7,8	7,8	7,8
Temperature (°C)	21,6	22,7	22,0
Sludge age (d)	18	18	18
Recycle ratios:			
1) from settler anoxic reactor	3:1	3:1	2:1
2) From first aerobic to anoxic reactor	0:1	3:1	3:1

Table 4.5a Average results of the bench scale unit (Fig 4.4) in the first series of tests (See Tables A3.1a to A3.6a in Appendix A_3 for daily results)

Parameter	I	A	II	III	E
O_2 upt.rate	-		85,2	21,9	-
COD	490	27,3	22,9	22,4	17,9
VSS	-	2117	2177	2064	2189*
TKN	47,6	10,0	5,5	1,7	2,0
NH_3	-	9,8	4,2	0,2	0,0
NO_3	0,0	1,2	5,6	9,8	9,2

I = influent
A = anoxic reactor
II = first aerobic reactor
III = second aerobic reactor
E = effluent
* = weighted average
Average temperature : 25,1°C

Table 4.5b *Average results of the bench scale unit (Fig 4.4) in the second series of tests from 27.10.76 to 9.11.76 (See Tables A3.1b to A3.6b in Appendix A_3 for daily results.)*

Parameter	I	A	II	III	E
O_2 up	-		87,1	16,0	-
COD	461	30,0	20,8	18,4	15,4
VSS	-	2473	2456	2386	2444*
TKN	46,1	7,6	4,3	1,7	2,5
NH_3	-	5,0	0,7	0,1	0,0
NO_3	0,1	3,3	7,2	8,9	8,9

I = influent
A = anoxic reactor
II = second aerobic reactor
E = effluent
* = weighted average
Average temperature : 23,8°C

Table 4.6a Average values of the measured parameters in the pilot plant unit in the first series of tests from 5.10.76 to 20.10.76. (See Tables A4.1a to A4.6a in Appendix A_4 for daily results)

Parameter	I	A	II	III	IV	V	E
O_2 upt.rate	-	-	59,3	36,5	23,5	19,3	34,6
COD	477	24,9	19,0	17,5	18,2	17,5	15,7
VSS		2550	2447	2466	2406	2477	2469*
TKN	45,1	9,2	4,8	3,3	2,6	2,0	1,9
NH_3		9,4	3,4	6,6	6,3	0,2	0,0
NO_3	0,3	1,2	5,8	9,2	9,7	9,8	8,7

I = influent
A = anoxic reactor
II = first aerobic reactor
III = second aerobic reactor
IV = third aerobic reactor
V = fourth aerobic reactor
E = effluent
* = average VSS
Average temperature : 21,6°C

Table 4.6b *Average values of the measured parameters in the pilot plant unit in the second series of tests from 21.10.76 to 9.11.76. (See Table 4.1b to A4.6b in Appendix A_4 for daily results)*

Parameter	I	A	II	III	IV	V	E
O_2 upt.rate	-	-	66,8	38,2	26,3	14,4	35,2
COD	461	26,9	19,9	17,9	18,9	19,6	18,5
VSS		2609	2490	2501	2500	2496	2519*
TKN	46,6	8,3	6,3	3,4	2,6	2,4	2,0
NH_3		7,7	4,3	0,0	0,0	0,0	0,0
NO_3	0,1	1,7	4,3	7,7	8,2	8,4	7,4

I = influent
A = anoxic reactor
II = first aerobic reactor
III = second aerobic reactor
IV = third aerovic reactor
E = effluent
* = average VSS
Average temperature : 22,7°C

The average value for m_{xv} considering both series of tests in each of the two systems was m_{xv} = 3,19 mg VSS. mg COD.$^{-1}$d^{-1}. The average temperature for all the experiments was T = 23,3^{o}C i.e. b_{hT} = 0,264 d^{-1}. For these values the fraction f_{up} can be calculated using Eqs (4.52) : f_{up} = 0,04 mg VSS. mg COD^{-1}. This value is within the range of 0,02 to 0,05 usually encountered for settled sewage and equal to that found from data for settled sewage reported by Heideman (1979) (Fig 2.15).

Having estimated the values of the influent COD fractions f_{ca}, f_{us} and f_{up}, the only constant that still needs to be evaluated is the maximum specific growth rate for nitrifiers. This constant is most conveniently determined by carrying out trial and error simulations until the μ_{nmT} value is found that best simulates the observed nitrification behaviour.

In Figs 4.4 and 4.5 average experimental and corresponding simulated data for oxygen uptake rate, filtered COD, VSS, TKN and NO_3 concentration in the reactors and effluent of the bench scale system are shown plotted for the first and second series of tests respectively. It may be noted that there is good correspondence between all the measured parameters in each of the reactors for both series of tests. Similarly in Figs 4.6 and 4.7 the average experimental and corresponding simulated data of the measured parameters in the reactors and the effluent of the pilot scale system are shown plotted. Again good correspondence between experimental and simulated data may be observed.

The optimal value of μ_{nm20} (assuming an Arrhenius temperature dependency of θ = 1,123) was 0,33 mg X_n.mg X_n^{-1}.d^{-1} for both the bench and the pilot scale systems and in every aerobic reactor. From the fact that the same value for μ_{nm20} applied to all aerobic reactors in both systems to give good correspondence between predicted and observed data, it was concluded that the anoxic reactors did not have a measurable effect on nitrification behaviour in the aerobic reactors.

It is of value to compare the denitrification behaviour predicted by the general model (i.e. by computer simulation) with the behaviour predicted by the empirical model presented in Chapter 2, Eq (2.54). The extent of denitrification calculated by means of the general model, and by means of Eq (2.54) respectively,

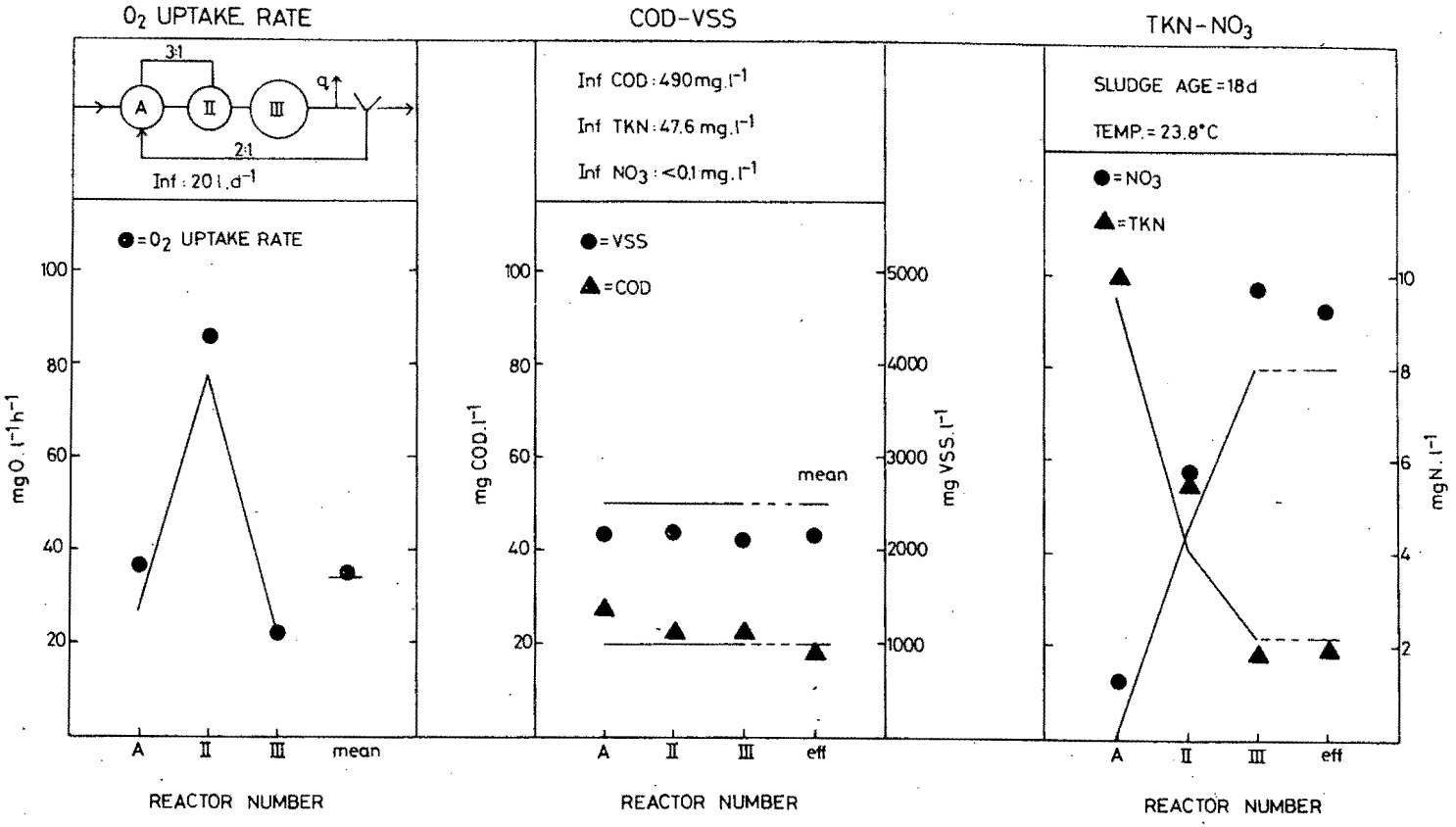

Fig. 4.4 *Experimental and simulated response of the bench scale system during the first series of tests under constant flow and load conditions. (Plant design parameters in Table 4.3)*

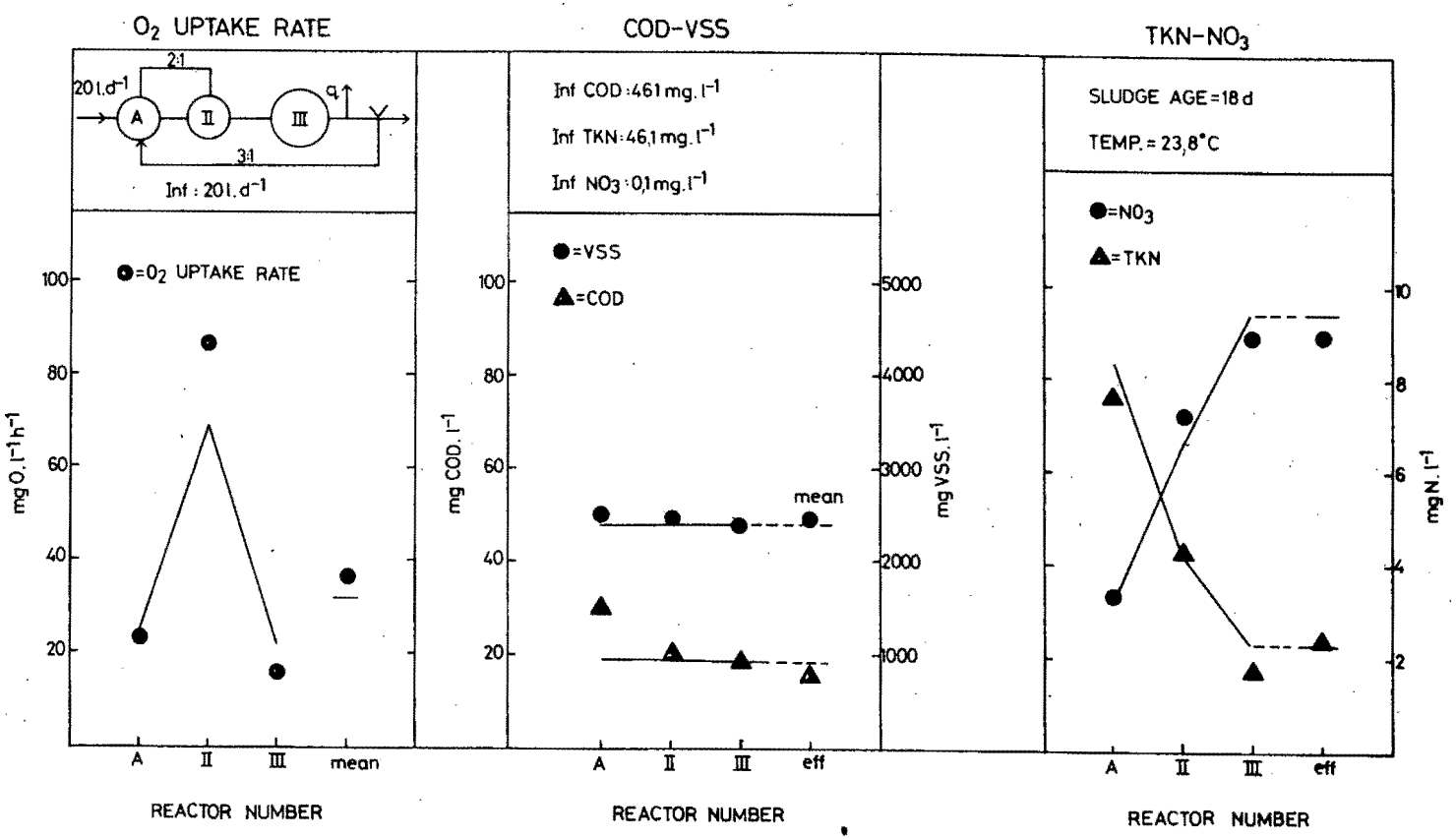

Fig. 4.5 *Experimental and simulated response of the bench scale system during the second series of tests under constant flow and load conditions. (Plant design parameters in Table 4.3)*

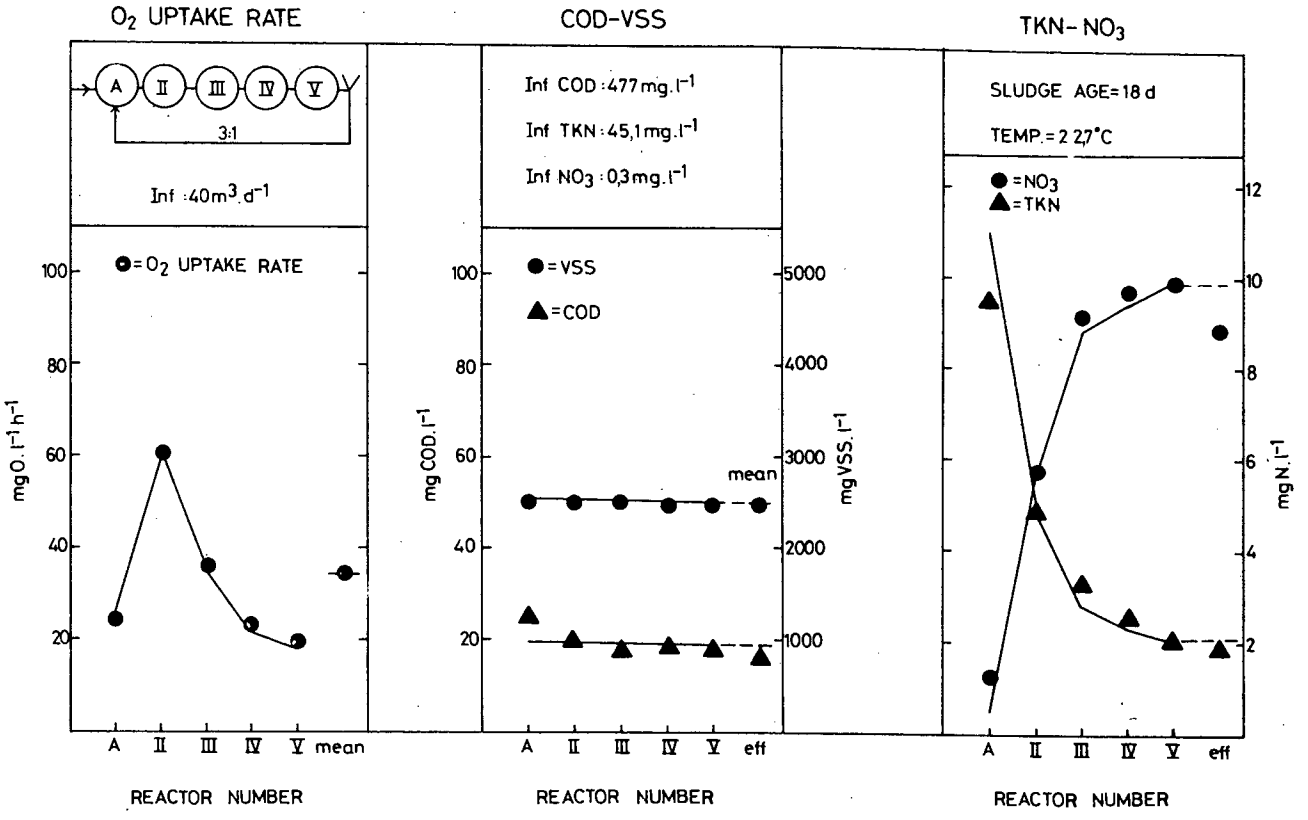

Fig. 4.6 *Experimental and simulated response of the pilot scale system during the first series of tests under constant flow and load conditions. (Plant design parameters in Table 4.4)*

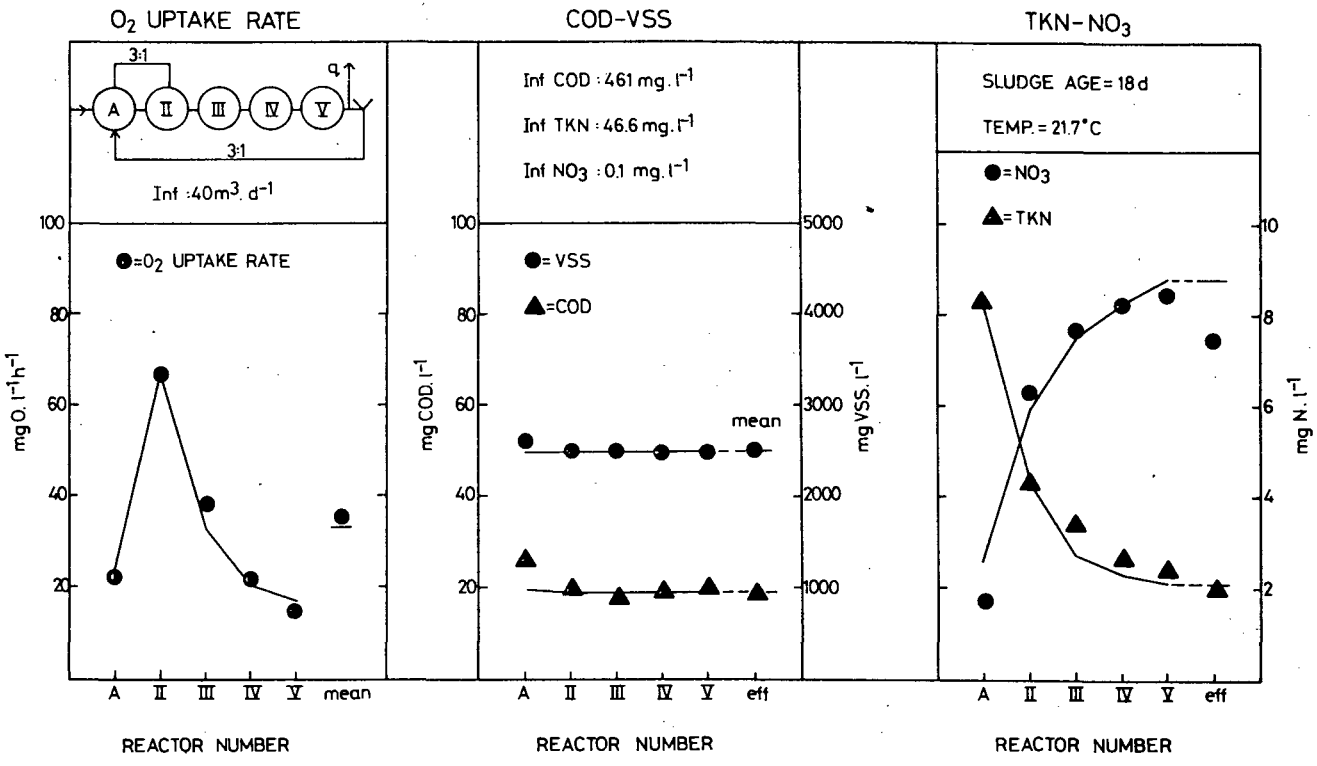

Fig. 4.7 *Experimental and simulated response of the pilot scale system during the second series of tests under constant flow and load conditions. (Plant design parameters in Table 4.4)*

are listed in Table 4.6a for the two series of tests on the bench scale system and the two series of tests on the pilot scale system respectively. It may be noted that in the first series of tests on the bench scale system both the general model and the empirical model predict complete denitrification in the anoxic reactor because the denitrification potential exceeds the nitrates available for denitrification. (The denitrification potentials predicted by the general and the empirical models were 29,2 and 29,8 mg $N.\ell^{-1}$ respectively, whereas the nitrate available for denitrification calculated from Eq (2.57) is only 28,9 mg $N.\ell^{-1}$). Consequently, in theory, nitrate removal in the anoxic reactor should be complete. Experimentally, however, a small residual nitrate (or nitrite) concentration was measured in the anoxic reactor (1,2 mg $N.\ell^{-1}$ - see Table 4.5a). In the second series of tests on the bench scale unit and in the two series on the pilot scale unit,the denitrification capacity was smaller than the nitrates available for denitrification and hence Eq 2.54 could be used to calculate the theoretical extent of denitrification using the empirical model.

From Table 4.6a the theoretical extent of nitrate removal calculated using the empirical model is always slightly higher than that calculated using the general model, except if complete denitrification in the anoxic reactor is predicted, in which event both are the same. The reason for the small difference is that the calculation using the empirical model assumes complete removal of the easily biodegradable material in the anoxic reactor, whereas in the general model a small residual concentration of easily biodegradable material always remains due to Monod kinetics characteristics and the remaining material does not contribute to nitrate removal. The differences between the predicted nitrate removals are so small that under constant flow and load conditions the empirical model can be used with confidence instead of the general model.

To compare the theoretical response of the systems to the experimental values, it is necessary to discuss how the experimental extent of denitrification can be estimated. As shown in Chapter 2 Section 2.5.6.1,there are basically two ways of calculating the experimental system nitrate removal:

Table 4.6a Theoretical system nitrate removal in the bench scale and the pilot scale system for both test series.

Experiments	Theoretical system nitrate removal		
	General Model	Empirical Model	Mean
Bench scale series I	28,9*	28,9*	28,9*
Bench scale series II	26,1	27,2	26,6
Pilot scale series I	25,6	26,4	26,0
Pilot scale series II	25,6	26,3	26,0

* Theoretically denitrification is complete in the anoxic reactor.

(1) By performing a nitrate balance over the anoxic reactor i.e. by using Eq (2.59).

(2) By performing a nitrogen balance over the whole system i.e. by using Eq (2.64).

In Table 4.6b the average experimental system nitrate removal based on both approaches as well as the mean of the two approaches are presented for the two systems and for the two series of tests carried out on each system. The two approaches do not give identical results: the experimental extent of nitrate removal using the nitrogen balance tends to be larger than that using the nitrate balance. Two possible reasons for this discrepancy can be identified:

(1) In the measurement of the oxidized nitrogen concentration only the sum of NO_2 + NO_3 was determined. It has been observed often that in an anoxic reactor there is an increase of the nitrite concentration of about 0,5 - 1,0 mg NO_2-N.ℓ^{-1}. In such a case if only NO_2 + NO_3 is determined this increase is not taken into account. It was shown in Section 3.5.6.1 of Chapter 2 that an increase of 1 mg NO_2-N.ℓ^{-1} is

Table 4.6b *Experimental system nitrate removal; averages of the bench scale and pilot scale system for both test series on the basis of nitrate and of nitrogen balances.*

Experiments	Experimental system nitrate removal		
	Nitrate balance	Nitrogen balance	Mean
Bench scale series I	29,5	28,8	29,2
Bench scale series II	25,5	28,5	27,0
Pilot scale series I	23,0	26,0	24,5
Pilot scale series II	26,5	27,8	27,2

equivalent to a reduction of 0,4 mg NO_3-N.ℓ^{-1} to nitrogen gas so that with the recycles used by Ekama (a = 3; s = 2 to 3) an increase of 1 mg NO_2-N in the anoxic zone is equivalent to about 2,5 mg NO_3-N.ℓ^{-1} influent. It can be concluded therefore that the *equivalent* system nitrate removal in the anoxic reactor may be up to 2,5 mg NO_3-N.ℓ^{-1} greater than the value indicated in Table 4.6b, when using the nitrate balance.

(2) When a nitrogen balance over the whole system is done it is assumed that all the denitrification takes place in the anoxic reactor but from the experimental data it is evident that some degree of denitrification takes place in the settler as well: the average nitrate concentration in the settler taking all experiments into account is about 0,5 mg NO_3-N.ℓ^{-1} lower than in the last aerobic reactor. With an "s" recycle ratio of 3:1 this signifies that the maximum extent of denitrification was about 1,5 mg NO_3-N.ℓ^{-1} in the settler.

Because the system nitrate removal of the anoxic reactor is probably slightly under-estimated when a nitrate balance is used and slightly over-estimated when a nitrogen balance is used, it is likely that a close approximation of the "real" experimental extent of denitrification can be obtained by taking the average of the two calculated values, as

shown in Table 4.6b; in all cases the average of the two values for the system denitrification approaches very closely the theoretically predicted nitrate removal.

Comparing Tables 4.6 (a and b) it can be concluded that the experimental extent of denitrification is closely predicted by both the empirical and the general models. Another conclusion is that the behaviour of the bench scale unit was very similar to that of the pilot plant; also that the low nitrate concentration in the anoxic reactors did not appear to have any detrimental effect on nitrification or denitrification response of the two plants.

3.2 Space dependent-time dependent systems

3.2.1 Pilot scale experiments

A second investigation using the same pilot plant as described in the previous section was carried out by Ekama and Marais (1976) under space and time dependent conditions. This was done by imposing a daily cyclic load as follows: a constant influent flow was drawn directly from the discharge of a primary settling tank of a full scale works, so that the natural diurnal variations of the COD, TKN and ammonia concentrations of the waste flow was imposed on the process. In order to keep the daily load the same as in the experiments under constant flow and load conditions (when the influent feed was drawn off from the primary settler when the concentration was maximum) it was necessary to increase the daily influent flow from 40 to 60 $m^3.d^{-1}$. The operational conditions for the cyclic load experiments are in Table 4.4

In Fig 4.8a the experimental values of influent COD and TKN load and of the oxygen uptake rates, TKN, NO_3, VSS and COD concentrations in the reactors are shown plotted as a function of time; in Fig 4.8b the corresponding simulated values are shown. Comparing Fig 4.8a and 4.8b there is good correspondence between the experimental and simulated data, but the correspondence is not so close as for constant flow and load conditions (Figs 4.6 and 4.7).

The imposition of cyclic flow and load conditions on a series reactor system exerts severe demands on the predictive power of

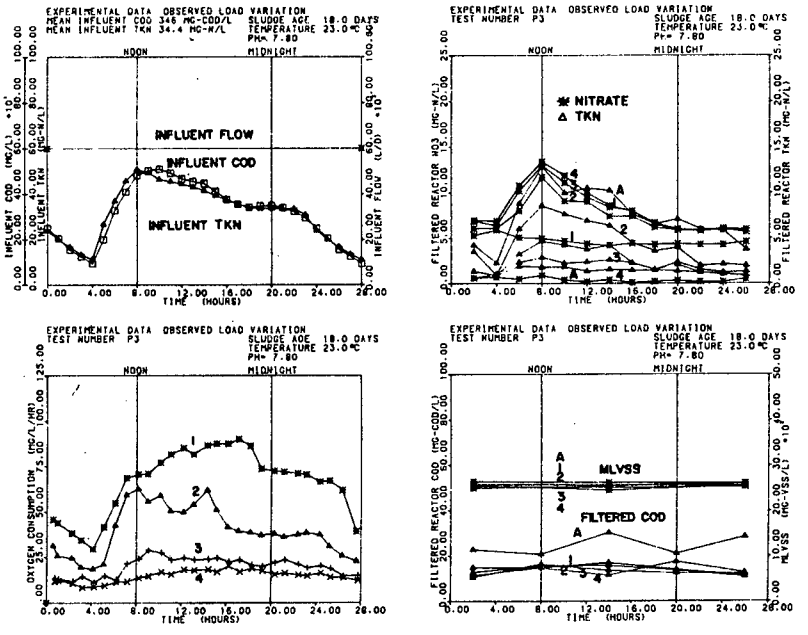

Fig. 4.8a *Experimental response.*

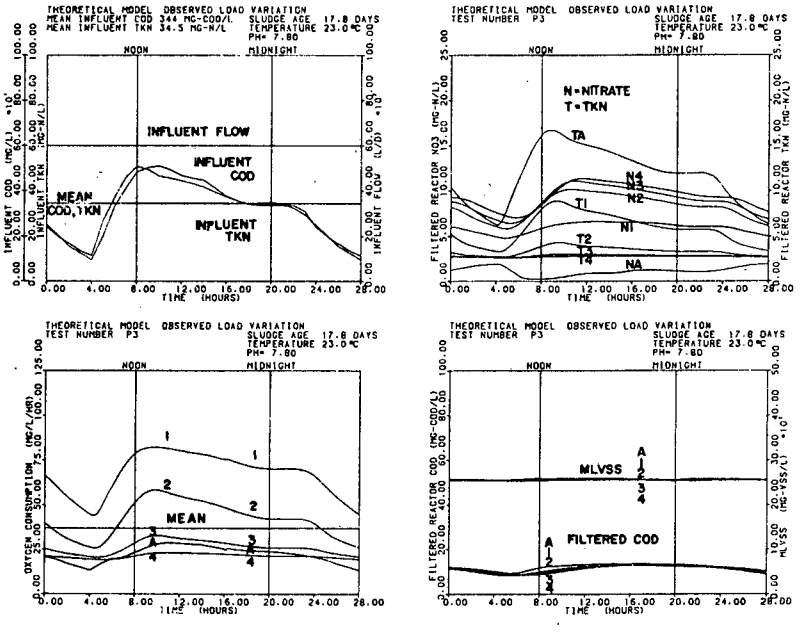

Fig. 4.8b *Simulated response.*

Fig. 4.8 *Experimental and simulated response of an anoxic/aerobic series reactor system under cyclic load conditions(configuration shown in Fig.4.7; plant design parameters in Table 4.4.*

the model and from a theoretical point of view this procedure should be the most satisfactory for checking the validity of the general model. Unfortunately the quasi plug flow conditions in the series system tend to cause unstable response if perturbations are present either in the values of the kinetic constants or of the flow and load conditions. This effect can be overcome only if the cyclic test is repeated a number of times and the results are averaged but the magnitude of work involved in such a number of repetitions makes this approach impractical.

3.2.2 Bench scale experiments

At laboratory scale, Wilson and Marais (1976) tested a two reactor process under space and time dependent conditions by imposing a very severely cyclic flow and load pattern. The process was a pre-denitrification configuration having an anoxic reactor (1,0ℓ) in series with an aerobic reactor (6,4ℓ). The underflow recycle ratio was 1,5:1 with respect to the mean daily influent flow. A square wave flow and load pattern (12 hours constant feed, 12 hours no feed) was imposed with a flow of $15\ell.d^{-1}$ of settled municipal sewage from the Cape Town-Athlone outfall ; this waste flow contains a high proportion of industrial discharge. In Table 4.7 the sewage characteristics and design parameters of the process are given for a 26-hour run. The experimental and simulated (solid lines) values of the TKN and NO_3 concentration in both reactors and of the oxygen uptake rate in the aerobic reactor are given in Fig 4.9. The average values of the effluent COD and VSS concentrations are also indicated. To obtain the best correlation between the experimental and simulated data it was necessary to assume that the fraction of easily biodegradable material was only 10 per cent of the biodegradable influent COD concentration (instead of the average fraction of 24 per cent reported by Ekama and Marais, 1978) and that the maximum specific substrate utilization rate constant for slowly biodegradable material had to be reduced from 3,0 to 2,3 mg COD. mg $VSS^{-1}.d^{-1}$, still keeping $K^1_{mp} = 0,38\ K_{mp}$. In addition a very low maximum specific growth rate of nitrifiers, $\mu_{nm20} = 0,17\ d^{-1}$, had to be used to describe the nitrification behaviour. With these adaptions it was possible to simulate quite closely the experimental data (see Fig 4.9).

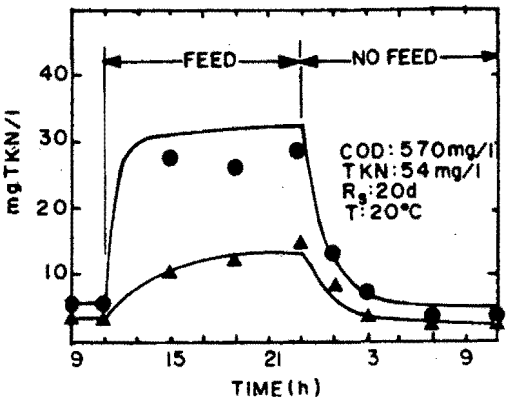

Fig. 4.9a *TKN - time profile.*

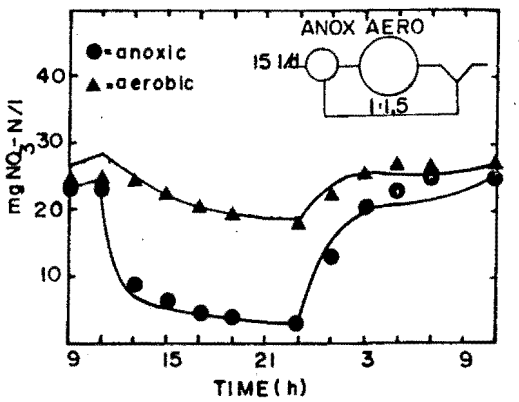

Fig. 4.9b *NO_3 - time profile.*

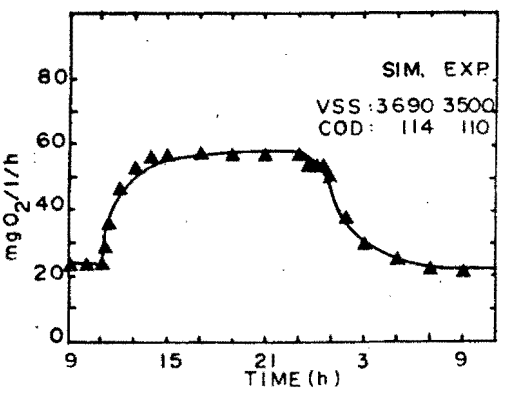

Fig. 4.9c *Oxygen uptake rate - time profile.*

Fig. 4.9 *Experimental and simulated response of TKN, NO_3 and O_2 uptake rate in a Modified Ludzack Ettinger system under cyclic (square wave) flow and load conditions.*

Table 4.7 Design parameters and sewage characteristics of the bench scale process operated by Wilson and Marais.

Parameter	Test
Configuration	Modified Lutzack Ettinger
Anoxic reactor	1,0ℓ
Aerobic reactor	6,4ℓ
Sludge Age	20d
Temperature	$20^{o}C$
Recycle ratio	1,5
Influent flow	$15\ell.d^{-1}$
Influent COD	570 $mg.\ell^{-1}$
Influent TKN	54,0 $mg.N.\ell^{-1}$
μ_{nm20}	0,17 d^{-1}

The worst correspondence between simulated and experimental data occurred just after the feed period was initiated or terminated, and it was found that a better fit could be obtained by slight modification of the recycle ratio indicating that hydraulic effects tended to distort the results. However the correspondence between simulated and experimental data was much better than in the case of the five reactor systems (i.e. quasi plug flow) operated under cyclic load conditions. This observation points towards the conclusion that, although theoretically it is advantageous to have many reactors (i.e. to have an hydraulic regime approach the plug flow regime), in practice a more stable and therefore more meaningful experimental response is obtained when the number of reactors is small.

Due to the relatively stable experimental response obtained

in a sewage inhibit the nitrification rate. Concomitant lower values for organic substrate utilization rate constants should also not be unexpected. If one accepts that the general model is reasonable then Wilson and Marais' data in fact provides the first positive evidence that the values of at least some kinetic constants depend on the sewage characteristics.

Accepting that the kinetic constants are dependent on the influent sewage characteristics, the problem devolves to what system will optimally provide estimates of the constants i.e. what system will require the least effort and yet allow the determination of constants with a high degree of accuracy and precision? Determination of the constants along the lines of the experiments described in Section 3 implies an extensive investigation and is not practical from a design point of view - there is a need for a simple experimental procedure by means of which estimates of the constants can be determined readily, accurately and precisely.

5. REQUIREMENTS OF A SYSTEM FOR INVESTIGATIONS INTO NITRIFICATION-DENITRIFICATION BEHAVIOUR

Requirements a system should fulfil in order to obtain reliable values for the kinetic constants of nitrification and denitrification as readily as possible can be listed as follows:

(1) Hydraulic effects should be minimized. Square wave cyclic flow and load conditions in particular, impose different hydraulic effects at the commencement and termination of the load cycles, the very regions critical to the determination of the constants.

(2) For accurate estimates of the kinetic constants it is desirable that the concentration of the pertinent variable (for example the nitrate concentration) should show as large a change as possible in time and/or space due to reaction alone. This theoretically can be achieved by having (a) plug flow reactors, or a high number of reactors in series, or operation under batch type of condition, (b) high influent concentration of

the reactants in nitrification and denitrification (i.e. high concentrations of N_{ti} and S_{bi} respectively) and (c) short sludge ages to produce a highly active sludge per unit sludge mass.

(3) A stable experimental response must be obtained i.e. the estimations of the kinetic constants must be reproduceable. This requirement tends to exclude plug flow reactors and series systems with a large number of reactors - mixing conditions in these systems are difficult or impossible to control, side wall effects often dominate, recycles need to be controlled very accurately to obtain good balances and some measurements such as the oxygen uptake rate are either impossible or very difficult to carry out accurately (Marais and Ekama, 1978).

(4) Denitrification in the settler must be avoided. From Figs 4,6 and 4.7 it can be noted that in the experiments carried out by Ekama and Marais (1976) the effluent nitrate concentration in the last reactor was higher than in the effluent, so that some denitrification must have taken place in the settler. However it is not possible to determine accurately the extent of this denitrification, and therefore the system nitrate removal in the anoxic reactor cannot be calculated reliably. This is reflected in the difference in calculated experimental system nitrate removal in Table 4.6b based on a nitrate balance or a nitrogen balance respectively. Denitrification in the settler can be minimised by a short retention time i.e. by a high underflow recycle ratio. A high dissolved oxygen concentration in the last reactor also will reduce the possibility of denitrification in the settler. However, often there is an internal recycle from the last aerobic reactor (which has the highest nitrate concentration) to an anoxic reactor and in this case a high dissolved oxygen concentration in the aerobic reactor is undesirable as the introduction of oxygen in the anoxic reactor might tend to reduce the rate of denitrification.

The best solution is to have a large settler so that the overflow velocity is low and the mass of sludge in the settler at any time is small.

A reactor system that largely satisfies all the requirements for a process to obtain reliable data on constants related to nitrification and denitrification is the single completely mixed reactor, alternately subjected to anoxic and aerobic periods and operated under constant flow and load conditions. By suitably selecting the time periods under anoxic and aerobic conditions, large changes in nitrate and ammonia concentrations can be induced while the influent flow and load are kept constant. By changing the proportion of time under aerobic and anoxic conditions and the actual lengths of these, the influence of the anoxic (or aerobic) environment on the kinetics of nitrification and denitrification can be established.

6. SINGLE REACTOR SEQUENTIALLY ANOXIC AND AEROBIC SYSTEM

Experiments on the single sequentially anoxic and aerobic system were carried out under constant flow and load conditions. The sludge mass was placed alternately in an anoxic and an aerobic environment by switching off the air supply over certain periods. A short sludge age of R_s = 6 days was chosen for all the experiments because this sludge age was sufficient to obtain efficient nitrification at the operational temperature (T = 20^{o}C) and yet would produce a sludge with a high fraction of active material, i.e. a sludge with a high reactivity per unit mass. The reactor volume was kept at 6ℓ i.e. for a sludge age of 6 days a total of 1ℓ of mixed liquor could be wasted daily from the system for the purpose of analysis of the different variables. The influent was raw sewage obtained from the Cape Town - Zeekoevlei outfall and stored at 4^{o}C. A batch of sewage was used for a period of 10 to 14 days. The raw sewage was diluted to a COD of 500 mg.ℓ^{-1} and the TKN was then increased to a value of approximately 50 mg. ℓ^{-1} by adding ammonium chloride. In order to ensure a stable pH, the alkalinity was increased to a value of about 300 ppm $CaCO_3$ by adding sodiumbicarbonate. The diluted sewage was

TABLE 4.8 Operational characteristics of the anoxic/aerobic single reactor system.

Reactor volume (l)	6
Sewage flow rate (l/d)	20
Hydraulic retention time (h)	7,2
Underflow recycle ratio	2:1
Actual retention time (h)	2,4
Sludge age (d)	6
Temperature (^{o}C)	20

kept at 4^{o}C and 20 ℓ was fed over a period of one day when a new dilution using the same batch of raw sewage was made. The contents of the feed tank was stirred slowly to prevent settling of the organic solids. The low temperature of the feed did not measurably affect the temperature of the mixed liquor in the reactor. The operational parameters which applied throughout the investigation are shown in Table 4.8.

A series of experiments was carried out having different sequences of anoxic and aerobic periods. For each experiment the process was run under the selected conditions for about a week before testing was started. During each experiment all the measureable parameters related to carbonaceous energy removal, nitrification and denitrification were determined as a function of time, i.e.

(1) Oxygen uptake rate (during aerobic periods)

(2) Filtered COD

(3) VSS concentration
(4) TKN concentration
(5) NH_3 concentration
(6) NO_3 concentration.
(7) Alkalinity
(8) pH

As about 100 ml of mixed liquor was required for the COD, VSS, TKN, NH_3, NO_3 and alkalinity tests, the daily volume of 1 ℓ of mixed liquor that had to be wasted from the reactor was withdrawn in 10 samples of 100 ml each. The sampling times were chosen such that these were expected to give maximum information about the process behaviour. Immediately after sampling,mercuric sulphate was added to the sample to kill the live bacteria and the sample was centrifuged and filtered. After filtration an immediate analysis of the alkalinity and ammonia concentration was carried out. This was done to avoid any change of alkalinity and ammonia concentrations due to possible hydrolysis of organic nitrogen. In addition determinations of influent and effluent COD, TKN, NH_3 and NO_3 concentrations, alkalinity and pH were done.

The experimental investigation was divided into two parts:

(1) One anoxic period per day : Anoxic periods of 2, 3, 4 and 5 hours per day were investigated. The objective of this part of the investigation was to determine the influence of the length of the anoxic period on nitrification and denitrification kinetics. Three tests were carried out for each of the anoxic periods of 2, 3 and 4 hours and four tests for the anoxic period of 5 hours.
(2) Cyclic anoxic and aerobic environments : In this part of the investigation, cycles of anoxic periods : aerobic periods of 1h : 2h or

of 2h : 2h were imposed on the series of experiments. The objectives of this part of the investigation were to impose very severe conditions for nitrification and denitrification in order to test the adequacy of the general program, and to determine the influence of the fraction of time the sludge was in an anoxic envoronment on the kinetics of the process. Three tests were carried out for a 1h : 2h alternating anoxic - aerobic environment and three tests for a 2h : 2h alternating anoxic-aerobic environment.

In total 19 experiments were carried out: 13 in the first part and 6 in the second part of the investigation. Appendix A_5 lists all the experimental data obtained.

In order to check that the alternating anoxic/aerobic single reactor system did not respond qualitatively differently from a "normal" anoxic/aerobic series reactor system, the response of the single reactor system was compared to the response of three Modified Ludzack Ettinger systems operated by Arkley and Marais (1981) during the same period and using the same sewage.

Simulation of the response of the single reactor system was carried out using the unsteady state program listed in Appendix A_2. To obtain estimates of the starting values for the variables the procedure set out in detail in Section 10.3.2 of Chapter 3, was followed. The simulated and experimental process response was plotted by a computer. The plotting program used is listed in Appendix A_6.

The high alkalinity of the feed (±300 ppm C_aCO_3) buffered the system so effectively that differences in mixed liquor pH during the anoxic and aerobic periods were inconsequential (see Fig 3.4 Chapter 3). Small increases in pH were observed during anoxic periods (from an average pH = 7,2 during aerobic periods to pH = 7,4 to 7,5 at the end of the anoxic period, depending on its length).

Between pH 7 to 8 such small changes in pH will have no effect on the kinetic constants and in the simulation it was assumed that the pH in the mixed liquor was constant and equal to the average measured pH i.e. no corrections were considered necessary for the influence of the time varying pH on the values of the kinetic constants.

Figures 4.10 to 4.13 show the experimental and simulated values of the measured variables (except pH) for experiments from the series in which there was only one anoxic period per day, for anoxic periods of 2, 3, 4 and 5 hours respectively. (Tables A5.1, A5.4, A5.7 and A5.11 in Appendix A_5 list the corresponding experimental values). Figure 4.14 shows the experimental and simulated response of an experiment with a cyclic anoxic/aerobic environment with sequential anoxic and aerobic periods of 1 and 2 hours respectively. Figure 4.15 shows the experimental and simulated response for a similar experiment but with anoxic and aerobic periods of 2 and 2 hours respectively. (The experimental data for Figs 4.14 and 4.15 are listed in Tables A5.14 and A5.17 of Appendix A_5). The experimental and simulated responses plotted in Figs 4.10 to 4.15 are representative examples of the total of 19 experiments carried out. The results of all the experiments are shown plotted in Figs A5.1 to A5.19 in Appendix A_5.

6.2 Discussion

6.2.1 Values of kinetic and other constants

The simulations in Figs 4.10 to 4.15 and in Appendix A_5 indicated that all the constants could be kept unaltered from those found by Dold et al (1980) for an aerobic environment (Table 3.9) or those listed in Table 4.2 for an anoxic environment, except for the following:

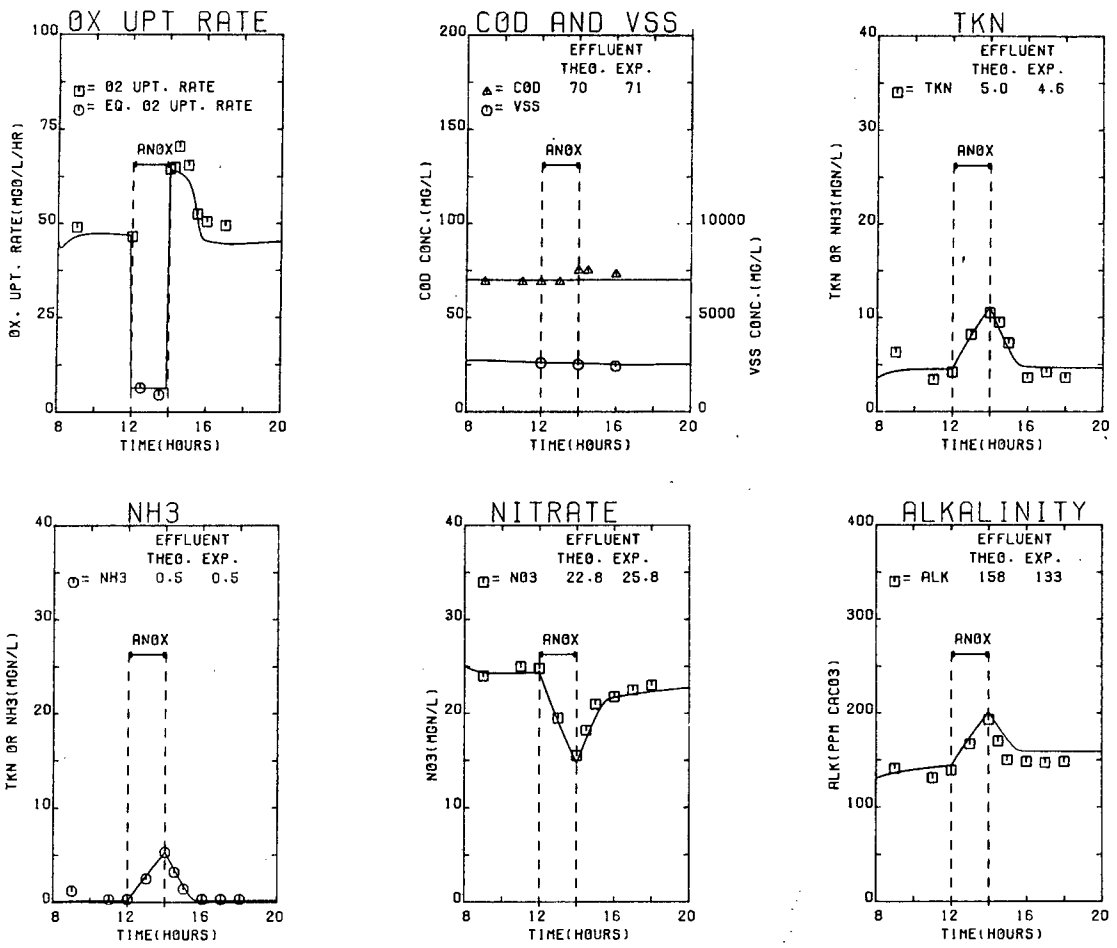

Fig. 4.10 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 2 hours per day. (Experimental data in Table A5.1)*

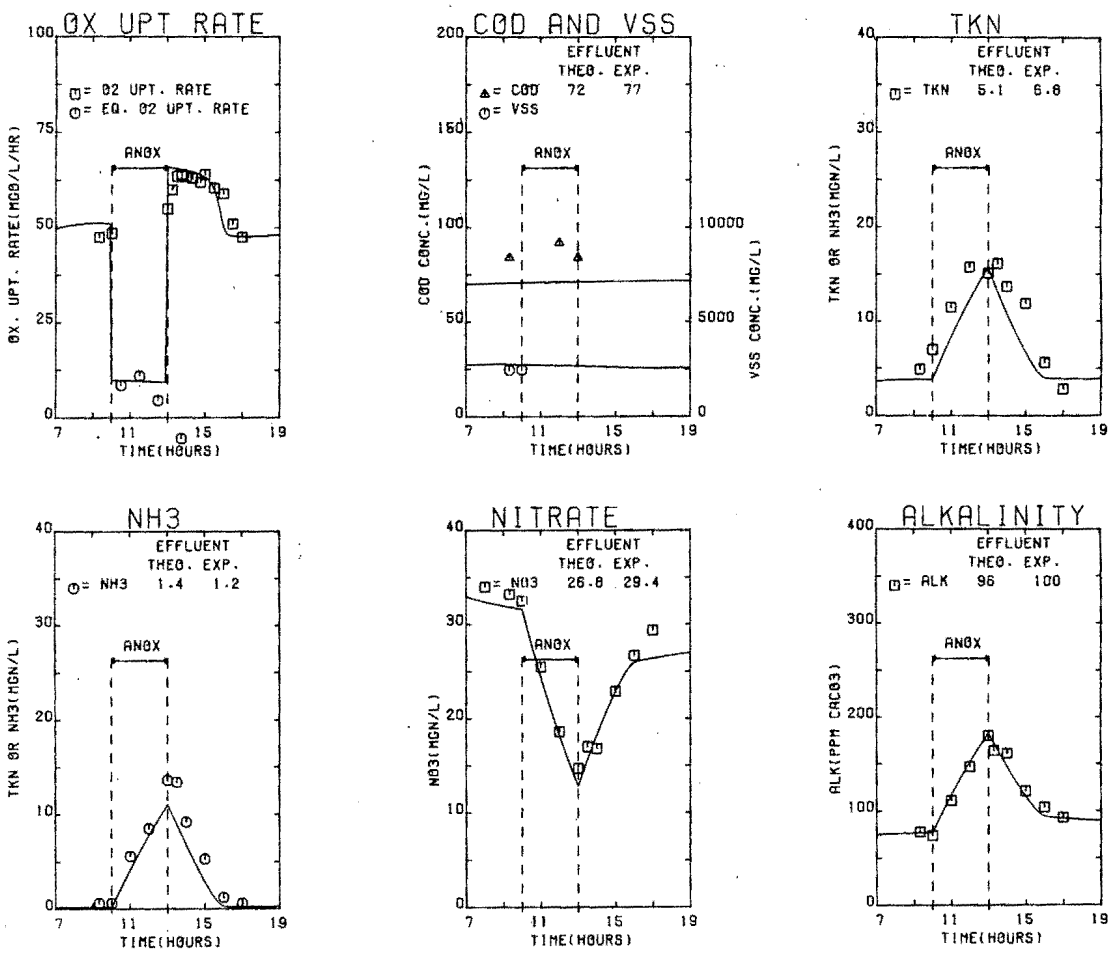

Fig. 4.11 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 3 hours per day. (Experimental data in Table A5.4)*

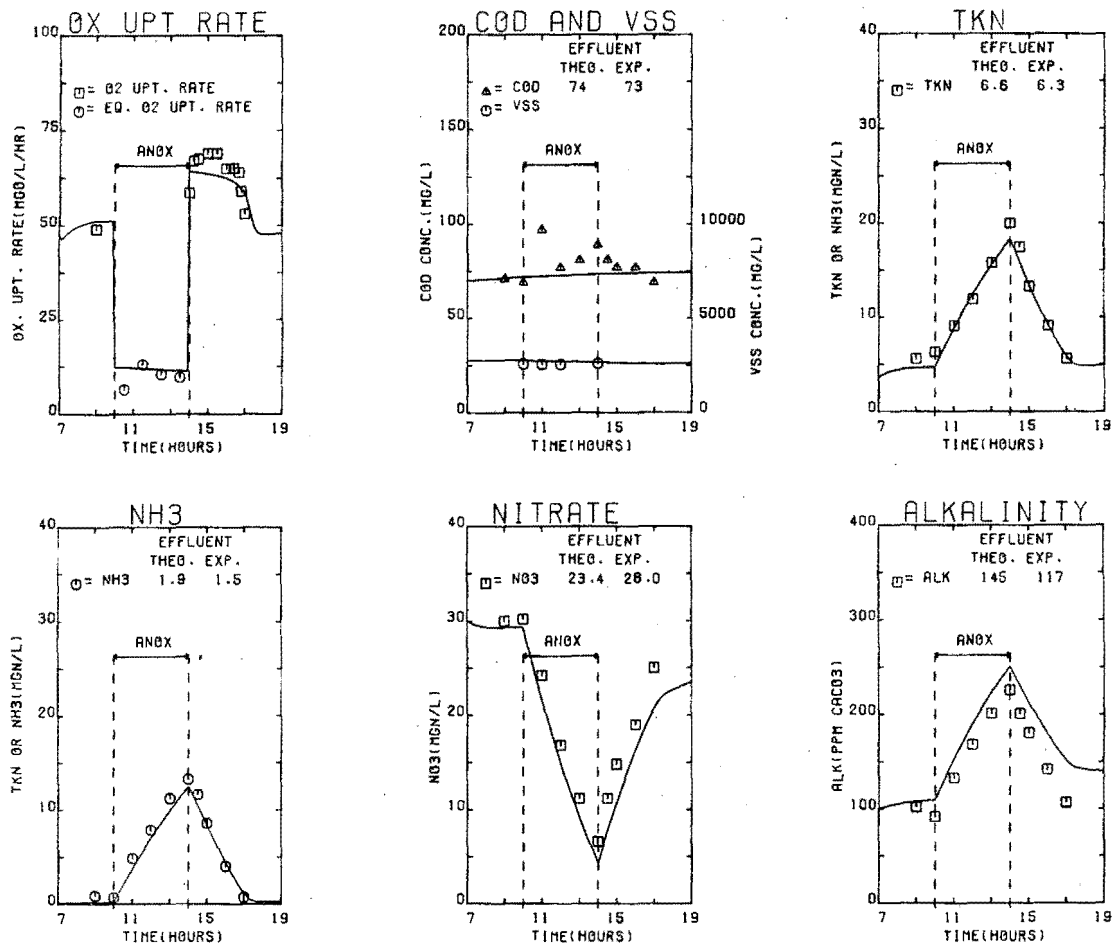

Fig. 4.12 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, ammonia and nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 4 hours per day (Experimental data in Table A5.7).*

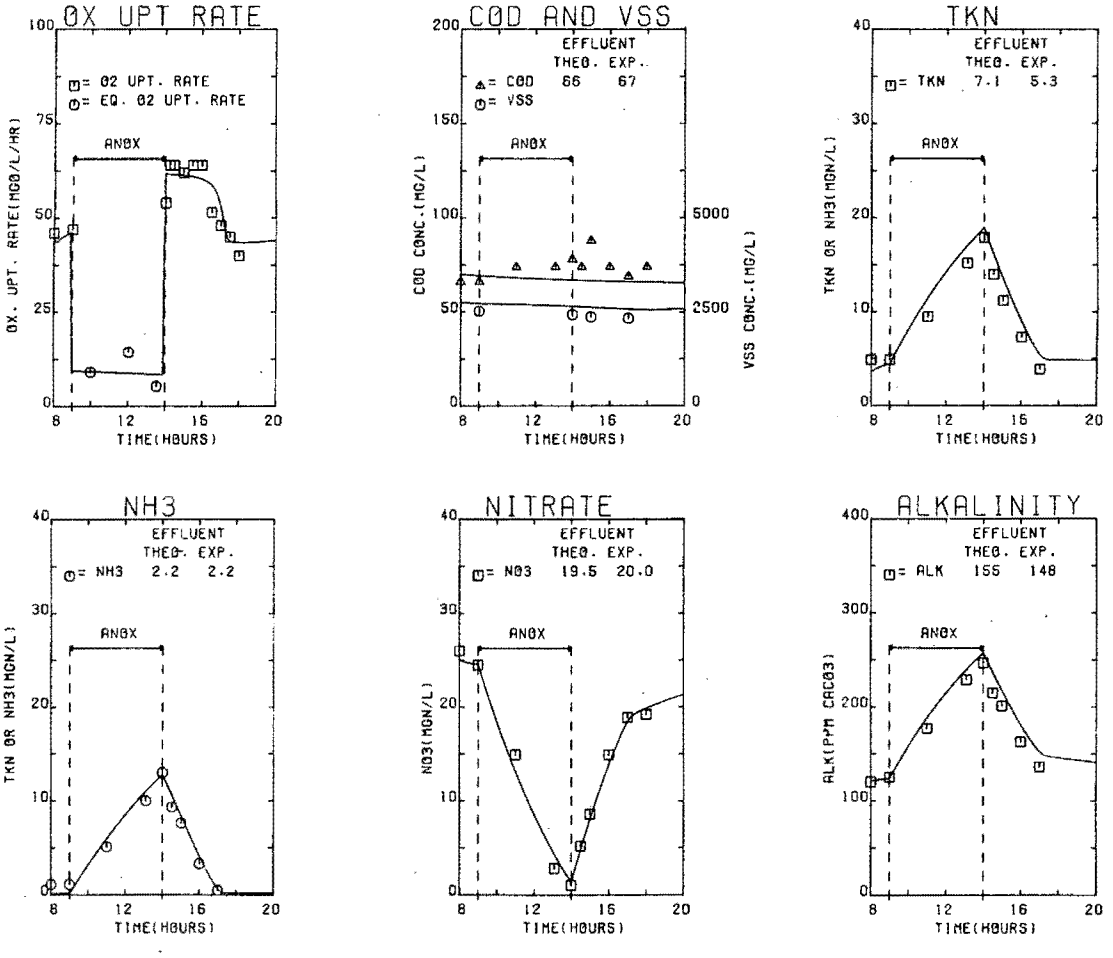

Fig.4.13 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 5 hours per day. (Experimental data in Table A5.11)*

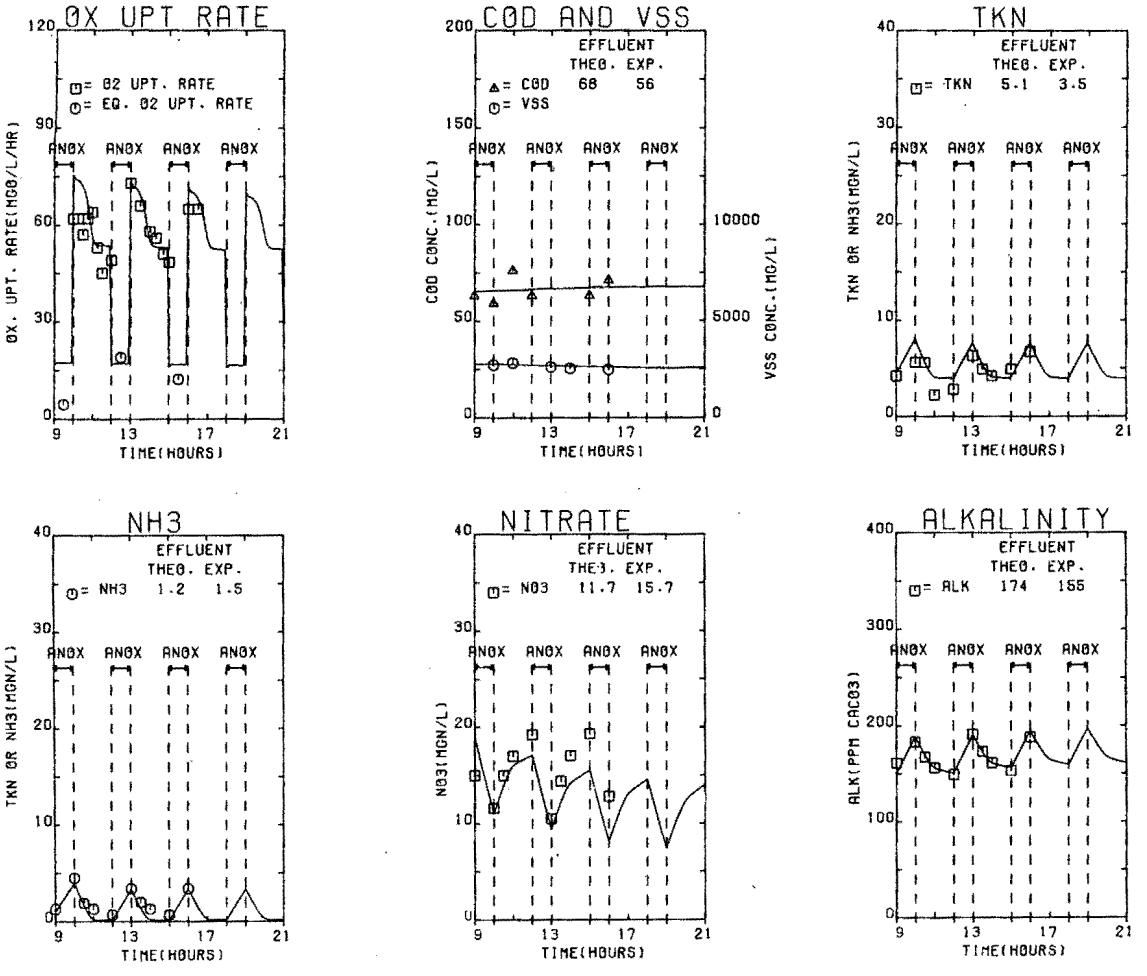

Fig. 4.14 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with alternating one hourly anoxic and two hourly aerobic periods. (Experimental data in Table A5.14)*

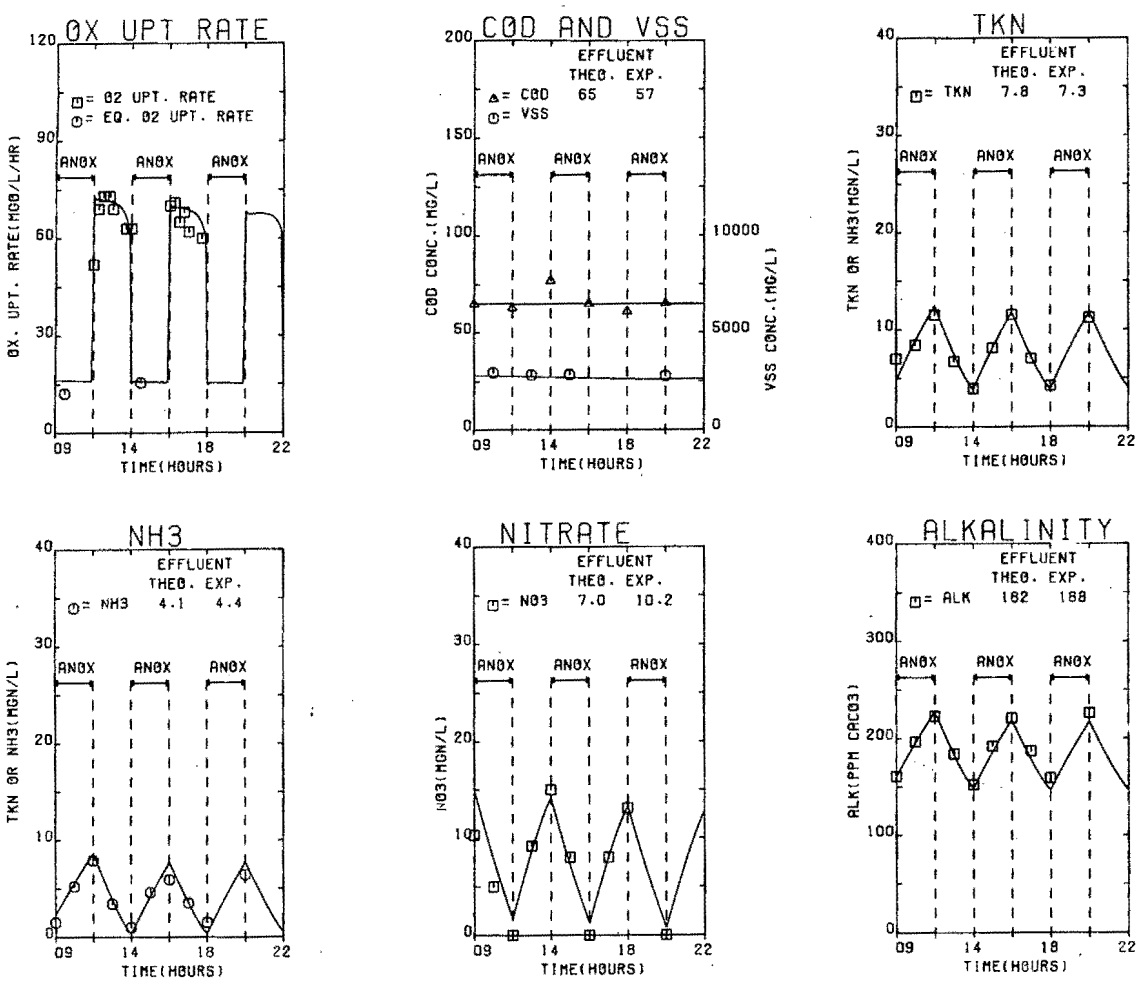

Fig. 4.15 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with alternating two hourly anoxic and two hourly aerobic periods. (Experimental data in Table A5.17)*

(1) f_{ca} = ratio of (easily biodegradable : total biodegradable COD in the influent). In all the experiments it was found that the best simulation was obtained for $f_{ca} = 0,05$ instead of $f_{ca} = 0,24$ as suggested by Dold, Ekama and Marais.

(2) K_{mp} = substrate utilization rate constant for slowly biodegradable material in an aerobic environment. The value of K_{mp} ranged from 2,2 to 2,5 instead of $K_{mp} = 3,0$ mg COD. $mgXa^{-1}.d^{-1}$ found by Dold *et al* (1980)

(3) μ_{nm2O} = Maximum specific growth rate of the nitrifiers. This value was found to be $\mu_{nm2O} = 0,45$ mg X_n.mg $X_n^{-1}.d^{-1}$ in all experiments, compared to values ranging from $\mu_{nm2O} = 0,17$ to $0,66\ d^{-1}$ in the previous experiments.

In addition it was found that in nine of the experiments the ratio $K^1_{mp} : K_{mp}$ differed from 0,38, the value calculated from the experimental results of Stern and Marais (1974) and Marsden and Marais (1976) (Figs. 4.2 (a and b)) and found applicable to the data by Sutton et al (1979) and Ekama and Marais (1976). However, it was found that in all experiments carried out with the same batch of sewage only one set of kinetic constants was required for optimal simulation of the experimental results of all the tests done with that particular batch. In Table 4.9 the values of K_{mp}, K^1_{mp} and the ratio K^1_{mp}/K_{mp} for optimal correlation between experimental and simulated data are presented for the different experiments in the chronological order in which these were carried out.

Table 4.9 *Values of K_{mp} and K^1_{mp} for the various experiments.*

Dates of experiments (1980)	Experiments number	Anoxic period (h)	K_{mp}	K^1_{mp}	K^1_{mp}/K_{mp}
14-16 May	4,5,6	3	2,2	0,66	0,30
28-30 May	7,8,9	4	2,2	0,84	0,38
5-8 June	10,11,12,13	5	2,2	0,66	0,30
10-12 June	1,2,3,	2	2,2	0,66	0,30
23-25 June	14,15,16	1:2*	2,5	0,95	0,38
26,27,30 June	17,18,19	2:2**	2,5	0,95	0,38

* cyclic with anoxic:aerobic period = 1:2 hours

** cyclic with anoxic:aerobic period = 2:2 hours

The value of f_{ca} could have been established conclusively by operating a completely mixed single reactor aerobic system under square wave flow and load conditions at a sludge age of 2,5 days as suggested by Ekama and Marais (1978). This was not done unfortunately, as such experiments had shown previously that f_{ca} had consistently a value of 0,24. Unbeknown to the laboratory, modification to the sewer reticulation system during this period caused qualitative changes to the waste characteristics. Subsequently when the effect of these changes became apparent, tests to determine f_{ca} were undertaken, using the Ekama-Marais procedure. In these experiments it was verified that f_{ca} indeed was significantly less than its originally assumed value of 0,24; f_{ca} values ranging from 0,05 to 0,10 were measured consistently.

Besides the indications above that the sewage characteristics were different from those in previous investigations,additional evidence was available. It was possible to verify that the low rates of substrate utilization and denitrification were not due to the alternating anoxic-aerobic mode of operation: During the time of the investigation described above, an experimental investigation was carried out by Arkley and Marais (1981) using three modified Ludzack-Ettinger systems with different anoxic sludge mass fractions. The same sewage was used in both investigations. The denitrification behaviour in these three systems during this period also was very different from the theoretical predictions using the kinetic constants developed from the experimental investigations by Marais and his co-workers. However, when the kinetic constants obtained from the simulations of the experiments with the single anoxic-aerobic reactor were inserted in the model,there was an excellent correspondence between observed and theoretical denitrification in each of the three systems. It was concluded, therefore, that the unusually low values of the constants f_{ca} and K_{mp} were due to atypical sewage characteristics during the period of the experimental investigation and could not be ascribed to the experimental set up using a single reactor anoxic-aerobic system.

6.2.2 Effect of the anoxic period on nitrification

From the nitrification behaviour the close correspondence between the experimental and simulated nitrate profile in Fig 4.13 implies that the imposed anoxic period of up to five hours did not appear to have any noticeable influence on the nitrification behaviour, i.e. no lag phase was apparent at 20°C. Apparently the rate of nitrification attained its maximum value almost immediately after the imposition of aerobic conditions. This conclusion is further substantiated by the good correspondence between

simulated and experimental profiles of TKN, ammonia, oxygen uptake rate and alkalinity during the aerobic period.

6.2.3 Alkalinity measurements

In Figs 4.10 to 4.15 and in the figures in Appendix A_5 it is apparent that good correspondence between observed and theoretical alkalinities is obtained when it is assumed that the mixed liquor alkalinity is affected only by (1) nitrification (2) denitrification and (3) nett ammonification. The effect of nett ammonification on alkalinity was very much smaller than the effects of nitrification and denitrification. An approximated value for the change in alkalinity can be calculated by considering the nett extent of ammonification:

$$N_{am} = N_{oi} - N_o - N_s \qquad (4.87a)$$

where N_{am} = ammonified TKN (mg $N.\ell^{-1}$ influent)

N_{oi} = influent organic nitrogen (mg $N.\ell^{-1}$ influent)

N_o = effluent organic nitrogen (mg $N.\ell^{-1}$ influent)

N_s = TKN (mg $N.\ell^{-1}$ influent)

In the experiments N_{oi} ranged from about 15 to 20 mg $N.\ell^{-1}$; the values of N_o and N_s were about 2 and 12 mg $N.\ell^{-1}$ respectively. Hence the nett extent of ammonification was the order of 1 to 6 mg $N.\ell^{-1}$, corresponding to alkalinity increases of 3,57 times larger, i.e. 3 to 20 mg $CaCO_3.\ell^{-1}$ influent. This range of values is very small when compared to the observed change in alkalinity in the system of about 150 to 200 mg $CaCO_3.\ell^{-1}$, depending on the extents of nitrification and denitrification. Hence as a first approximation the alkalinity changes in the mixed liquor can be ascribed to nitrification or denitrification. These alkalinity changes can be used to calculate approximate values of the rates of nitrification or

denitrification. A more accurate estimate of these rates can be obtained by assuming a constant rate of nett ammonification as indicated in Eq (4.87a) and by correcting the observed alkalinity changes for the effect of ammonification before carrying out the calculations to determine the rates of nitrification or denitrification. A manual procedure to calculate the rates of nitrification and denitrification from observed data is dealt with in the following Section 7.
In all the experiments of this investigation when the nitrification and denitrification rates were calculated on the basis of changes in the alkalinity (corrected for net ammonification), the rates were virtually identical to those obtained from the changes in the nitrate concentration. Hence it would appear that alkalinity measurements (when suitably corrected for ammonification) can substitute or complement nitrate concentration measurements in determining the rates of nitrification and denitrification.

6.2.4 System stability and reproduceability

Even though the changes of all the measured variables (except the VSS and filtered COD concentrations) were substantial it was noted from the results obtained on consecutive days that reproduceability in experimental response was very good. Thus only a small number of experiments were needed to obtain reliable experimental data. The precision of the values of the constants obtained by simulation for any particular experiment also was very high. This was indicated by the change in response simulated for a change in the value of a kinetic constant in the expressions for nitrification or substrate utilization - the simulated response was found to be very sensitive to the value assigned so that it was possible to obtain very precise values. In one instance errors were indicated by the simulations : In Fig 4.12 the rather poor correspondence between the observed and theoretical profiles of nitrate and alkalinity appeared to be due to a wrong estimation of the influent TKN. If the influent TKN is increased

by 5 mg $N.\ell^{-1}$ an almost perfect fit of experimental and theoretical profiles and of the experimental and theoretical average effluent concentrations of both nitrate and alkalinity is obtained, indicating that the determination of the influent TKN was in error.

From the theoretical and experimental behavioural characteristics of the anoxic-aerobic alternating single reactor system it would appear possible not only to obtain reliable experimental data and reproduceable values, but also experimental errors can be detected. In addition, and perhaps more important, the method has the advantages that it is relatively simple to operate and that much experimental data can be accumulated over a short period of operation. Overall, it constitutes a most convenient and time saving method of determining the kinetic constants for substrate utilization, nitrification and denitrification.

The general conclusion, taking all the work presented in Chapter 2 and this Chapter into account, is that the model characterized by the differential equations Eqs (3.86 to 3.95) for an aerobic environment and Eqs (4.13 to 4.22) for an anoxic environment gives a good description of all the nitrification- denitrification systems, with the proviso that the numerical values of some of the kinetic constants are affected by the characteristics of the influent sewage. The constants that seem to be most sensitive to changes in influent sewage characteristics are the maximum specific growth rate of the nitrifiers (reflected in the maximum nitrification rate) and,to a lesser extent,the maximum specific utilization rate constant for slowly biodegradable material (reflected in the utilization rate of dissolved oxygen and the denitrification rate). Also the fraction of easily biodegradable material in sewage may be subject to considerable variations. This implies that the numerical values of the empirical denitrification rate "constants" K_2, K_3 and α also depend on the sewage characteristics and that accurate values for particular waste flow can only be determined by experiment, most conveniently

by utilizing the single reactor-constant flow and load sequential aerobic/anoxic procedure outlined above.

7. APPROXIMATE DETERMINATION OF NITRIFICATION AND DENITRIFICATION KINETICS WITHOUT THE AID OF A COMPUTER

If a digital computer to carry out simulations is not available, it is still possible to obtain approximate values for the kinetic constants of nitrification, denitrification and substrate utilization. If a computer is available this approximate solution in any case is useful as a first estimate of these kinetic constants in the simulation procedure. In order to obtain the approximate solution it is first necessary to investigate the relationship between the influent-effluent concentration-time profile and the reaction rate of a particular substance in a reactor.

7.1 Estimation of the reaction rate in a completely mixed reactor from influent-effluent concentration profiles

In a completely mixed reactor, j, if a reaction takes place involving a substance variable, C, then the rate of change of the concentration of that variable with time, r_{Cj}, is a function of the reaction rate, r_{Cjr}, and the hydraulic loading rate, r_{Cjh}. (See Chapter 3, Section 10).

$$r_{Cj} = r_{Cjr} + r_{Cjh} \tag{4.88}$$

The rate of change of the variable C in reactor j due to the hydraulic effect can be determined by considering that with a constant reactor volume:

$$V_j r_{Cjh} = V_j \, (dC_j/dt)_h = Q_{ij} C_{ij} - Q_{ej} . C_{ej} \tag{4.89}$$

where r_{Cjh} = rate of accumulation of C in reactor j due to hydraulic loading

V_j = volume of reactor j

Q_{ij} = flow rate into reactor j

Q_{ej} = flow rate from reactor j

C_{ij} = concentration of C in influent to reactor j

C_{ej} = concentration of C in effluent from reactor j

= concentration in reactor j (CSTR)

= Cj

For a constant flow $Q_{ij} = Q_{ej} = Q_j$ the hydraulic retention time in reactor j is constant and given by

$$R_j = V_j/Q_j \tag{4.90}$$

Hence from Eq (5.89)

$$(dC_j/dt)_h = (C_{ij} - C_j)/R_j \tag{4.91}$$

Eq (4.88) can now be rewritten as:

$$r_{cj} = (dC_j/dt) = r_{Cjr} + r_{Cjh}$$

$$= r_{Cjr} + (C_{ij} - C_j)/R_j \tag{4.92}$$

Hence:

$$\frac{dC_j}{r_{Cjr} + (C_{ij} - C_j)/R_j} = dt \tag{4.93}$$

For the special case of a constant reaction rate $r_{Cjr} = r_r$ and a constant influent concentration C_{ij} the differential equation Eq (4.93) can be solved:

$$(C_{ij}-C_j)/R_j + r_r = \left[(C_{ij}-C_j/R_j + r_r \right] e^{-t/R_j} \tag{4.94}$$

From Eq (4.94) it may be noted that the constant reaction rate r_r can be determined by maintaining C_{ij} constant for a long period to allow the exponential term on the right hand side to die away (for $t > 3R_j$ = the value of $e^{-t/R_j} < 0{,}05$). The reaction rate r_r is

now readily determined by

$$r_r = (C_j - C_{ij})R_j \qquad (4.95)$$

However in the activated sludge process reaction rates may change considerably over periods much shorter than the hydraulic retention time of the reactor and the approach above is of no value for the determination of the reaction rate. In this case an approximated value for the reaction rate at any time t may be calculated as follows:

For a variable rate r_{Cjr} from Eq (4.88):

$$r_{Cjr} = r_{Cj} - r_{Cjh}$$

$$= (dC_j/dt) - (dC_j/dt)_h \qquad (4.96)$$

where (dC_j/dt) = observed rate of change of C in reactor j

$(dC_j/dt)_h$ = rate of change of C in reactor j due to hydraulic loading

During a short interval Δt between, say, times t_1 and t_2 the observed change in the concentration C_j is, say, from C_{jt1} to C_{jt2}. The rate of change can be approximated by linearization:

$$r_{Cj} = (dC_j/dt) = \Delta C_j/\Delta t = (C_{jt2}-C_{jt1})/(t_2-t_1) \qquad (4.97)$$

The rate of change of the concentration C_j due to hydraulic effects over the same period $\Delta t = t_2 - t_1$ can be approximated by taking the arithmetic mean value of the concentration in the reactor as the effluent concentration over that period. With Eq (4.93)

$$(dC_j/dt)_h = (C_{ij} - \overline{C_j})/R_j \qquad (4.98)$$

where $\overline{C_j} = (C_{jt1} + C_{jt2})/2$

= average concentration in the reactor over the period from t_1 to t_2

Both Eqs (4.97) and (4.98) are linearizations and the approach above is only valid if the period Δt is short so that the relative change from C_{jt1} to C_{jt2} (i.e. $(C_{jt1} - C_{jt2})/C_{jt1}$) is small.

Small variations in the influent concentration C_{ij} can be accommodated by taking the average influent concentration over the period t_1 to t_2. The hydraulic loading rate can now be expressed as:

$$(dC_j/dt)_h = (\overline{C_{ij}} - \overline{C_j})/R_j \qquad (4.99)$$

where $\overline{C_{ij}} = (C_{ijt1} + C_{ijt2})/2$

$\overline{C_{ij}}$ = average concentration of C in the influent to reactor j

C_{ijt1} = influent concentration of C in reactor j at time t_1

C_{ijt2} = influent concentration of C in reactor j at time t_2.

An approximated value of the average reaction rate r_{cjr} over the period t_1 to t_2 in reactor j can now be calculated by substitution of Eq (4.97) and (4.98) in (4.99) :

$$r_{Cjr} = (C_{jt2} - C_{jt1})/(t_2 - t_1) - (\overline{C_{ij}} - \overline{C_j})/R_j \qquad (4.100)$$

7.2 Estimation of the rates of nitrification and of denitrification

The application of Eq (4.100) will now be illustrated in the calculation procedure followed to determine the rates of nitrification and of denitrification. The pre-denitrification system operated by Wilson and Marais and described earlier in Section 3.2.2 (Table 4.7 and Fig 4.9) will be taken as an example. In a dynamic test such as carried out by Wilson and Marais, the rates of nitrification and denitrification changes with time but approximated values can be calculated by means of Eq (4.100). Columns 2 and 3 of Table 4.10 show the concentrations of nitrate in the anoxic and the aerobic reactor at the various times during the 26 hour test.

Table 4.10 NITRATE PROFILES AND ASSOCIATED RATES OF NITRIFICATION AND DENITRIFICATION FOR TEST 2 (WILSON AND MARAIS)

1	2	3	4	5	6	7	8	9	10	11
	Nitrate concentration		Observed rate		Average influent		Average effluent		Denitrification rate	Nitrification rate
Time	Anoxic	Aerobic	Anoxic	Aerobic	Anoxic	Aerobic	Anoxic	Aerobic		
*11.00	24,1	25,0								
12.00			-7,20	-0,15	10,7	15,9	15,9	24,9	-5,3	2,9
13.00	8,7	24,7								
14.00			-1,15	-1,1	10,1	7,5	7,6	23,6	-6,7	4,4
15.00	6,4	22,5								
16.00			-1,35	-0,95	9.2	5,1	5,1	21,6	-10,4	5,9
17.00	4,7	20,6								
18.00			-0,50	-0,45	8,6	4,2	4,2	20,2	-10,1	5,0
19.00	3,7	19,7								
20.00										
21.00			-0,33	-0,55	8,0	3,1	3,1	18,6	-11,00	4,7
22.00										
23.00**	2,4	17,5								

(continued)

Table 4.10 (continued)

1	2	3	4	5	6	7	8	9	10	11
	Nitrate concentration		Observed rate		Average influent		Average effluent		Denitrification rate	Nitrification rate
Time	Anoxic	Aerobic	Anoxic	Aerobic	Anoxic	Aerobic	Anoxic	Aerobic		
0.00			5,2	2,6	20,1	7,6	7,6	20,1	-6,5	4,4
1.00	12,8	22,7								
2.00			4,15	1,5	24,2	17,0	17,0	24,2	-2,6	2,6
3.00	21,1	25,7								
4.00			0,60	0,65	26,3	21,7	21,7	26,3	-3,7	1,3
5.00	22,3	27,0								
6.00			1,25	-0,50	25,5	23,5	23,5	25,5	-0,6	-0,2
7.00	24,8	26,0								
8.00										
9.00			-0,1	10,33	26,6	24,0	24.0	26.6	-2.4	0.7
10.00										
11.00	24,4	27,3								

* Start feed (15ℓ/d in 12 h) ** Stop feed

From these values the observed average rates of change in the nitrate concentration can be calculated for the intervals between the times the nitrate concentrations were determined, by using Eq (4.97). The values of the observed average rates of change in the nitrate concentrations are listed in Columns 4 and 5 of Table 4.10 for the anoxic and the aerobic reactor respectively.

The nitrate concentrations in both reactors change with time but the average concentration can be approximated over the interval between two measurements as:

$$\overline{C_j} = (C_{jt1} + C_{jt2})/2 \qquad (4.101)$$

This approximation is only valid if the relative change from C_{jt1} to C_{jt2} is not considerable with respect to C_{jt1} or C_{jt2}. The average influent concentrations to a reactor over the interval also can be calculated: Nitrate enters into the anoxic reactor via the underflow recycle, s (there is no 'a' recycle),where the nitrate concentration can be taken as equal to that in the aerobic reactor, i.e.

$$\overline{C_{i,1}} = \overline{C_2} \cdot Q_s/(Q_s+Q) \qquad (4.102)$$

where $\overline{C_{i,1}}$ = average nitrate concentration enetering the anoxic reactor over the interval

$\overline{C_2}$ = average concentration in the aerobic reactor over the interval

Q_s = flow rate from the settler

Q = flow rate of the influent

The average influent nitrate concentration into the aerobic reactor is equal to the average effluent concentration from the anoxic reactor, the latter in turn being equal to the average concentration in the anoxic reactor, i.e.

$$\overline{C_{i,2}} = \overline{C_1} \qquad (4.103)$$

where $\overline{C_{i,2}}$ = average nitrate concentration entering the aerobic reactor over the interval

$\overline{C_1}$ = average nitrate concentration in the anoxic reactor over the interval

The values of the average nitrate concentrations in the flows entering and leaving the reactors can now be calculated. In Table 4.10 the average influent and effluent nitrate concentrations for the anoxic and for the aerobic reactor are calculated from the nitrate profiles in Columns 2 and 3 and presented in Columns 6 to 9. These values allow the determination of the rate of change of nitrate due to hydraulic effects in both reactors using Eq (4.99).

The approximated values of the rates of denitrification in the anoxic reactor and of nitrification in the aerobic reactor can now be calculated. For the rate of denitrification Eq (4.100) is applied to the anoxic reactor, where the retention time is given by :

$$R_{1a} = V_{an}/(Q + Q_s)$$

$$= 0{,}457\text{h during feed period}$$

$$= 1{,}067\text{h during non-feed period} \qquad (4.104)$$

The approximated values of the rates of denitrification over the different periods of measurement, calculated with Eq (4.100) and using the numerical values in Columns 2 - 9 of Table 4.10 and R_{1a} from Eq (4.104) are shown in Column 10 of Table 4.10.

Similarly the rates of nitrification in the aerobic reactor can be calculated. The retention time in the aerobic reactor is given by :

$$R_{2a} = V_{aer}/Q + Q_s)$$

$$= 2,93 \text{ h during feed period}$$

$$= 6,83 \text{ h during non-feed period} \qquad (4.105)$$

The calculated values of the nitrification rates are in Column 11 of Table 4.10.

7.3 Estimation of kinetic constants for nitrification and denitrification

Although the rates of nitrification and denitrification calculated above are only approximated values, these are of great assistance in estimating the kinetic constants that are needed in the general program, for simulation of the experimental data. This can be shown as follows:

(1) Nitrification

During the feed period the rate of nitrification in the aerobic reactor is maximum because the ammonia concentration is high during this period (see Fig 4.9). From Table 4.10, column 11 it may be noted that the average nitrification rate during this period is 5,0 mg $N.\ell^{-1}.h^{-1}$. (The first value of 2,9 is excluded, because it is likely to be influenced by the very big change in influent nitrate concentration over the measuring period from 11 to 13 hours). The associated kinetic constant can now be estimated by considering

$$r_n = -(dN_a/dt)_n = (dX_n/dt)/Y_n = \mu_n X_n/Y_n \qquad (4.106)$$

Over the considered period the ammonia concentration is very high so that $\mu_n \simeq \mu_{nm}$ and

$$\mu_{nm} = \mu_n = Y_n r_n/X_n \qquad (4.107)$$

The concentration of the nitrifying organisms $_n$ may be estimated from the mean concentration of TKN nitrified in the system (Eq 3.206)

$$X_n = Y_n R_s \Delta N / ((1+b_{nT} R_s) R_h) \qquad (4.108)$$

In the experiment the mean concentration of nitrified TKN, ΔN, was estimated at 32 mg $N.\ell^{-1}$

i.e. $X_n = 72$ mg $VSS.\ell^{-1}$

The value of μ_{nm} can now be estimated from Eq (4.107):

$$\mu_{nm} = 0{,}1*5{,}0*24/72 = 0{,}17\ d^{-1} \qquad (4.109)$$

This is identical to the "best" value in the simulation with the general program.

(2) Utilization of biodegradable organic material

From the profile of the oxygen uptake rate in the anoxic period in Fig. 4.9 one can observe that during the feed period this parameter tends towards a constant value of about 60 mg $O.\ell^{-1}h^{-1}$. Under the peak loading conditions the nitrification rate in the aerobic reactor is about $r_n = 5{,}0$ mg $N.\ell^{-1}hr^{-1}$ which corresponds to an oxygen uptake rate for nitrification of $O_n = 4{,}57.r_n = 23$ mg $O.\ell^{-1}h^{-1}$ so that the oxygen uptake rate for carbonanceous material removal in the aerobic reactor under peak loading conditions is about $O_c = 60 - 23 = 37$ mg $O.\ell^{-1}hr^{-1}$. Due to the presence of a pre-denitrification reactor, which will remove most if not all the easily biodegradable material, it can be assumed that in the aerobic reactor the oxygen consumption for carbonaceous oxygen removal is associated with the utilization of slowly biodegradable material, X_s. With a ratio of oxygen consumption : organic substrate, utilization equal to $(1 - PY_h)$ the oxygen consumption rate O_c can be related to the

rate of utilization of slowly biodegradable material :

$$O_c = (1 - PY_h).r_{up}$$

$$= (1 - PY_h).\ K_{mp}.X_sP/(X_sP + X_a.K_{sp}).X_a. \qquad (4.110)$$

Due to the fact that the oxygen consumption rate attains an almost constant value during the feed period it may be assumed that the value of X_s is so large that the ratio $X_sP/(X_sP + X_a\ K_{sp})$ is practically constant. This ratio is constant if $X_s.P >> X_a.K_{sp}$ in which case the ratio is about 1. Hence approximating

$$O_c \simeq (1 - PY_h).\ K_{mp}.X_a \qquad (4.111)$$

The value of X_a can be calculated readily from Eq (3.148) and Table 4.7 For the experiment under consideration $X_a = Y_hR_s.S_{bi}/((1 + b_{hT}\ R_s)R_h) \simeq 1250$ mg VSS.ℓ^{-1}. Hence an approximated value for K_{mp} can now be calculated as :

$$K_{mp} \simeq O_c/((1 - PY_h)X_a) = 2{,}1 \text{ mg COD.mg } X_a^{-1}.d^{-1} \qquad (4.112)$$

The correctness of the value of K_{mp} hinges around the error made by taking the ratio $X_sP/(X_sP + K_{sp}\ X_a)$ equal to 1. In fact this ratio is less than 1 and the value of K_{mp} will be correspondingly higher, but nevertheless, the calculated value may serve as a first trial value in the simulation of the experimental data. In the computer simulation the best value for K_{mp20} was found to be 2,2 mg COD.mg $X_a^{-1}d^{-1}$, which is only five per cent higher than the approximated value calculated above.

(3) Denitrification

If the assumption is made that the value of K^1_{mp} i.e. the utilization rate constant for slowly biodegradable material in the anoxic reactor is a fraction 0,38 of the value of K_{mp} (i.e.

the corresponding constant in the aerobic reactor) and also assuming $X_s P >> X_a . K_{sp}$ in the anoxic reactor, the rate of denitrification in the anoxic reactor due to utilization of slowly biodegradable material during the feeding period is given by Eq (4.73):

$$\begin{aligned}(dN_2/dt) &= -(1 - PY_h)\, r_{up}/2{,}86 \\ &= -(1 - PY_h)\, K^1_{mp}\, X_s P (X_s P + X_a . K_{sp}) . X_a/2{,}86 \\ &= -0{,}38.\ (1 - PY_h)\, K_{mp} . X_a/2{,}86 \\ &= -5{,}0 \text{ mg } NO_3 - N . \ell^{-1} hr^{-1} \end{aligned} \qquad (4.113)$$

From Table 4.10, during the feed period the denitrification rate is about 10 mg $NO_3 - N . \ell^{-1} hr^{-1}$, which means that the rate of denitrification due to utilization of easily biodegradable material is approximately 10-5 = 5 mg $N . \ell . hr$. Hence the rate of utilization of easily biodegradable material must be 5 x 2,86/(1-PY) = 42 mg $COD . \ell^{-1} hr^{-1}$. With an anoxic reactor volume of 1,0ℓ, this means that 42 mg of S_{bs} has to be introduced with the influent flow. During the feed period the influent flow rate is 1,25$\ell . h^{-1}$ so that the required concentration of easily biodegradable influent COD is about S_{bsi} = 42/1,25 = 34 mg $COD . \ell^{-1}$. This corresponds to about 10 per cent of the influent biodegradable COD concentration, i.e. $f_{ca} - S_{bsi}/S_{bi}$ = 0,10. This value was also found by simulation using the general model.

Thus in an experiment such as carried out by Wilson and Marais it is possible to calculate the kinetic constant for nitrification μ_{nmT} but in order to calculate the constants that determine the rates of denitrification, f_{ca} and K^1_{mp}, the assumption must be made that the ratio K^1_{mp}/K_{mp} is known (in this case K^1_{mp}/K_{mp} = 0,38 was assumed). If the easily biodegradable influent fraction is determined separately, this assumption is not necessary, in this case K^1_{mp} and K_{mp} can be evaluated independently from the nitrate and oxygen uptake rate profiles respectively.

7.4 Calculation procedures for single reactor systems

The procedure outlined above for the two reactor

system can also be applied to a single reactor alternating anoxic and aerobic system, and, if the fraction of easily biodegradable influent material, f_{ca}, is known the kinetic constants for nitrification and denitrification can be calculated. To illustrate the procedure, an analysis is done on the results from the last test carried out in the investigation using the single reactor system (Test 19 in Table A5.19). In Test 19 anoxic periods of two hours were alternated with aerobic periods of two hours. In Table 4.11a the observed nitrate concentration (Column 3) is listed as a function of time (Column 2) for the different samples (Column 1). From the changes in the nitrate concentration with time the observed rates of change of the nitrate concentration over the intervals between NO_3 determinations are calculated by means of Eq. (4.97) and are listed in Column 5. The observed rates must now be adjusted for hydraulic effects in order to obtain the rates due to reaction alone. The rates of change due to hydraulic effects (washout) is calculated by applying Eq. (4.99) and using the average nitrate concentration over each interval (listed in Column 4) and the results (i.e. the hydraulic rate effect, r_h) are listed in Column 6. The reaction rate , r_r, i.e. the rate of nitrification (during aerobic periods) or denitrification (during anoxic periods) are now calculated using Eq. (4.100) i.e. subtracting the hydraulic rate from the observed rate of change of nitrate concentration. Values for r_r are listed in Column 7. By averaging the denitrification rates (negative values) and the nitrification rates (positive values) respectively, the mean denitrification and nitrification are obtained: The mean rate of denitrification is:

$$r_{den} = (4,8 + 6,0 + 7,7 + 3,6 + 6,4)/5$$
$$= 5,7 \text{ mg N.}\ell^{-1}h^{-1}$$

and the mean rate of nitrification is:

$$r_n = (8,8 + 8,7 + 7,6 + 8,6)/4$$
$$= 8,3 \text{ mg N.}\ell^{-1}h^{-1}$$

Table 4.11a Nitrification and denitrification rates in Test 19 (Table A5.19, Appendix A_5) calculated on the basis of nitrate concentration-time profile

(1)	(2)	(3)	(4)	(5)	(6)	(7)
Sample	Time	NO_3^-	$\overline{NO_3}$	r_c	r_h	r_r
1	9.30*	12.5				
			9.5	-6.1	-1.3	-4.8
2	10.30	6.4				
			3.2	-6.4	-0.4	-6.0
3	11.30**	0.0				
			4.1	+8.2	-0.6	+8.8
4	12.30	8.2				
			11.4	+6.4	-1.6	+8.2
5	13.30*	14.6				
			10.0	-9.1	-1.4	-7.7
6	14.30	5.5				
			2.8	-4.0	-0.4	-3.6
7	15.30**	0.5				
			4.0	7.0	-0.6	+7.6
8	16.30	7.5				
			11.5	7.0	-1,6	+8.6
9	17.30*	15.5				
	18.30		8.0	7.5	-1.1	-6.4
10	19.30**	0.5				

* = start anoxic period

** = end anoxic period

The values of the mixed liquor alkalinity also can be used to calculate the rates of nitrification and denitrification as follows: First it is necessary to estimate the rate of change of alkalinity due to ammonification: In test 19 (Table A5.19 in Appendix A5) the influent organic nitrogen concentration N_{oi} is 49.0 - 33.5 = 15,5 mg $N\ell^{-1}$; the nitrogen concentration per litre influent wasted in the sludge (based on 10 per cent mass fraction of nitrogen in the VSS), N_s, is about 13,5 mg $N.\ell^{-1}$ and the effluent organic nitrogen, N_o, is about 3,5 mg $N.\ell^{-1}$. Hence there is a very small negative ammonification rate:

$$r_{am} = (N_{oi} - N_o - N_s)/R_h$$
$$= (15,5 - 3,5 - 13,5)/7,2$$
$$= -0,2 \text{ mg } N.\ell^{-1}h^{-1}$$

This rate is so small that it can be neglected so that the changes of the alkalinity in the mixed liquor can be attributed exclusively to hydraulic effects and nitrification or denitrification.

The calculation procedure to find the rate of change of alkalinity due to reaction alone is identical to that outlined above for nitrate and is shown in Table 4.11b. The average rate of increase of alkalinity due to denitrification during the anoxic periods is :

$$(r_{alk})_{den} = (14,4 + 16,2 + 27,6 + 16,6 + 20,8)/5$$
$$= 19,0 \text{ ppm } CaCO_3.h^{-1}$$

The rate of decrease of alkalinity due to nitrification during the aerobic periods is:

$$(r_{alk})_{nit} = (61,7 + 54,1 + 57,1 + 55,3)/4$$
$$= 57,0 \text{ ppm } CaCO_3.h^{-1}$$

Assuming the stoichiometric ratios for denitrification (1 mg NO_3-N $\equiv$ 3,57 mg $CaCO_3$) and nitrification (1 mg NO_3-N $\equiv$ 7,14 mg $CaCO_3$) to be valid, the rates of denitrification and nitrification based on the changes in alkalinity are calculated as follows:

Table 4.11b Nitrification and denitrification rates in Test 19 (Table A5.19, Appendix A_5) calculated on the basis of the alkalinity-time profile

(1)	(2)	(3)	(4)	(5)	(6)	(7)
Sample	Time	Alk	$\overline{Alk}$	r_C	r_h	r_r
1	9.30*	80				
			98	35	20.5	14.5
2	10.30	115				
			131	32	15.8	16.2
3	11.30**	147				
			124	-45	16.7	-61.7
4	12.30	102				
			86	-32	22.1	-54.1
5	13.30*	70				
			94	48	21.0	27.0
6	14.30	118				
			134	32	15.4	16.6
7	15.30**	150				
			130	-41	16.1	-57.1
8	16.30	109				
			92	-34	21.2	-55.3
9	17.30*	75				
	18.30		114	39	18.2	20.8
10	19.30**	153				

* = start anoxic period

** = end anoxic period

$$r_{den} = 19,0 \text{ ppm } CaCO_3.h^{-1}$$
$$= 19,0/3,57 = 5,3 \text{ mg } N.\ell^{-1}h^{-1}$$

$$r_n = 5,70 \text{ ppm } CaCO_3.h^{-1}$$
$$= 5,70/7,14 = 8.0 \text{ mg } N\ell^{-1}h^{-1}$$

The rates of nitrification and of denitrification based on both the nitrate-time and the alkalinity time profile are listed in Table 4.11c. Possibly an acceptable practice is to take the average rates as representative for the process, as neither one or the other variable implicitly has greater weight.

Table 4.11c Nitrification and denitrification rates based on nitrate concentration-time and alkalinity-time profiles

	Nitrification rate mg $N.\ell^{-1}h^{-1}$	Denitrification rate mg $N.\ell^{-1}h^{-1}$
Nitrate profile	8,3	5,7
Alkalinity profile	8,0	5,3
Average	8,15	5,5

From the average rates of nitrification and denitrification the kinetic constants now can be determined as follows:

(1) Nitrification: The μ_{nmT} value can be estimated by assuming that during aerobic periods the nitrification rate is maximum (which is reasonable in view of the high measured ammonia concentrations). In order to estimate μ_{nmT}, the concentration of Nitrosomonas must first be evaluated:

From Eq (3.206) :

$$X_n = \Delta N.Y_n.R_s /((1 + b_{nT} R_s)R_h)$$

The concentration of the nitrified TKN, ΔN, is estimated from the difference between the influent TKN, N_{ti} = 49 mg $N.\ell^{-1}$, the effluent TKN, N_{te} = 7,0 mg $N.\ell^{-1}$ and the nitrogen concentration for sludge wastage N_s = 13,5 mg $N.\ell^{-1}$; i.e.
ΔN = 49 - 7 - 13,5 = 28,5 mg $N.\ell^{-1}$ and X_n = 56 mg VSS ℓ^{-1}

Hence

$$\mu_{nm} = Y_n r_n / X_n = 0,43\ d^{-1}$$

(2) Denitrification:

Assuming f_{us} = 0,12 (From Table A.19) and f_{up} = 0,09 mg VSS. mg COD^{-1} the influent biodegradable COD concentration $S_{bi} = (1 - f_{us} - P.f_{up})\ S_{ti}$. = 370 mg $COD.\ell^{-1}$. For the estimated (or measured) value of f_{ca} = 0,05 the easily biodegradable material in the influent has a concentration of S_{bsi} = 0,05. 370 = 18,5 mg $COD\ell^{-1}$. With a flow of 0,83$\ell.h^{-1}$ this means that 15 mg S_{bs} enters the reactor per hour and causes a nitrate removal of 15 *$(1-PY_h)/2,86$ = 1,8 mg $N.h^{-1}$ i.e. the rate of denitrification due to utilization of S_{bs} is 1,8/V= 1,8/6 = 0,3 mg $N.\ell^{-1}h^{-1}$. Hence the rate of denitrification due to the utilization of slowly biodegradable material is about

(dN_2/dt) = 5,5 - 0,3 = 5,2 mg $N.\ell^{-1}h^{-1}$
= 125 mg $N.\ell^{-1}d^{-1}$

Now using Eq (4.113) the value of K^1_{mp} is estimated assuming $X_s P >> K_{sp}.X_a$:

$$K^1_{mp} = (dN_2/dt)/((1 - PY_h)X_a/2,86)$$

The estimated active sludge concentration of X_a = 1250 mg VSS.ℓ^{-1} (Eq 3.148), Y_h = 0,45 mg VSS.mg COD^{-1} and P = 1,48 mg COD. mg VSS^{-1}. Hence:

$$K^1_{mp} = 0{,}86 \text{ mg COD mg } X_a^{-1}.d^{-1}$$

The kinetic constants μ_{nm} and K^1_{mp} determined above by means of the approximated manual method (μ_{nm} = 0,43 d^{-1} and K^1_{mp} = 0,86 mg COD. mg $X_a^{-1}d^{-1}$) are close to the optimal values obtained from the computor simulations : μ_{nm} = 0,45 d^{-1} and K^1_{mp} = 0,96 mg COD. mg VSS$^{-1}d^{-1}$. The operational procedure used in test number 19 was particularly useful in determining the values of the kinetic constants because of two reasons:

(1) because of the cyclic mode of operation (2 hours anoxic period followed by 2 hours aerobic period) it was possible to obtain a large number of relevant determinations of the nitrate concentration and the alkalinity.

(2) due to the large anoxic fraction the utilization of the substrates ammonia and slowly biodegradable organic material was incomplete. This is indicated by the fact that during the aerobic periods the oxygen uptake rate was essentially constant, i.e. the values of O_c and O_n and hence also of r_{us}, r_{up} and r_n must have remained essentially constant during the aerobic periods. This can only be so if during the whole period $X_s P/(X_s P + K_{sp} X_a) \simeq 1$ and $N_a/(N_a + K_{nT}) \simeq 1$ so that the approximations that must be made to calculate K^1_{mp} and μ_{nm} were justified.

From the discussion above it would appear that a good method to calculate the kinetic constants for nitrification is the single reactor completely mixed systems placed cyclicly under an anoxic and an aerobic environment, proportioning the anoxic and aerobic periods such that during the aerobic periods,the utilization of both ammonia and of stored material is incomplete.

CHAPTER 5

OPTIMIZATION OF NITROGEN REMOVAL

1. INTRODUCTION

In the previous chapter a general model was presented describing the kinetics of nitrification and denitrification in single sludge activated sludge systems. While this model was shown to give a good account of the experimental process response under a wide range of reactor configurations and influent flow and load conditions, it does not provide information on the optimal design of the process, i.e., the configuration of anoxic and aerobic reactors, their relative volumes, recycles and recycle flow rates and operational sludge age, for maximum removal of nitrogen by biological denitrification in a particular design situation.

In developing an optimal design procedure, account must be taken of certain constraints that are operative on any single sludge nitrification-denitrification system. These are:

(1) At all times efficient nitrification must be guaranteed, say 95 per cent oxidation of the available ammonia to nitrate. This requirement, for any selected sludge age, places a constraint on the minimum fraction of sludge mass in the process that must be in an aerobic environment; consequently there is a corresponding maximum fraction of the sludge mass that can be in an anoxic environment.*

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* A second limitation is set on the maximum anoxic sludge mass fraction: As the anoxic sludge mass fraction increases beyond about 45 to 50 per cent, the whole process may be adversely affected; in particular the settleability of the sludge may be reduced severely (Arkley and Marais, 1981).

(2) The extent of denitrification that can be achieved in a particular anoxic reactor is dependent upon the relative size of the reactor and its location in a series configuration. The extent of denitrification possible is given by the denitrification potential of the reactor (See Chapter 2, Section 3.7).

(3) In a particular anoxic reactor, the maximum extent of denitrification will be developed fully only if the mass of nitrates conveyed to that reactor is equal to or greater than the denitrification potential, i.e., if the denitrification reaction is not limited by the availability of nitrate.

In any design procedure, cognisance must be taken of all these constraints.

A framework for optimal design cannot be established by considering the problem in its generality. Simplified conditions of flow and load must be assumed in order to incorporate the constraints above explicitly in a design procedure.**

The response of the final design under the expected cyclic flow and load conditions can always be checked using the general model. Then, by trial and error, adjustments of operational parameters (such as the ratio of aerated:unaerated sludge mass) it may be possible to obtain slightly greater nitrogen removals than that indicated by the "optimal" design based on the simplified flow and load condition. From experience, however, optimal design under constant flow and load conditions usually is also near optimal for cyclic flow and load conditions. In practice, taking due regard of the uncertainties that are always present in the expected influent flow and load patterns and the kinetic constants for nitrification and denitrification, such

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** In some instances it is possible to have empirical rules linking the simulated response of the process under cyclic flow and load conditions to the response under constant flow and load conditions. In those instances the design procedure for steady state conditions can be extended directly to include optimal design under variable flow and load conditions. See Section 6 of this Chapter.

adjustments to the theoretical optimum usually have little merit.

This Chapter will consider the development of a design procedure for optimal nitrogen in single sludge nitrification-denitrification systems, within the constraints set out above.

2. NITRIFICATION UNDER ANOXIC/AEROBIC CONDITIONS

The general theory of nitrification in anoxic/aerobic systems developed in the previous chapter is founded on three basic assumptions:

(1) Nitrifying organisms, i.e. Nitrosomonas, being obligate aerobes, can grow only under aerobic conditions.

(2) Death of nitrifiers takes place under both anoxic and aerobic conditions.

(3) Nitrification kinetics under aerobic conditions is not affected by the presence of anoxic reactors.

The general model developed on the basis of these assumptions was shown to simulate nitrification behaviour very well, both in anoxic/aerobic series reactor systems (Figs 4.4 and 4.5) and, in alternating anoxic/aerobic single reactor systems (Figs 4.10 to 4.15). Under constant flow and load conditions nitrification behaviour can be described in good approximation with simple expressions and these form the basis for design.

2.1 Approximate formulation for nitrification under constant flow and load conditions

Under constant flow and load conditions, an explicit expression for the effluent ammonia concentration from an anoxic/aerobic series reactor system can be derived if two approximations are accepted:

(1) There is a uniform concentration of nitrifying organisms throughout the series reactor system. In reality there is

growth of nitrifiers only in the aerobic reactor while death takes place in both anoxic and aerobic reactors; hence the concentration of nitrifiers in an aerobic reactor should be slightly higher than in a preceding or a subsequent anoxic reactor. However, from simulations by means of the general model, the differences in concentration of nitrifiers between different reactors of a series system are so small that these may be neglected.

(2) The effluent concentration of ammonia is equal to the ammonia concentration in the aerobic reactor. Strictly speaking this is true only in pre-denitrification systems (Fig 5.1a) where the aerobic reactor is the last one in a series. In systems with a post-denitrification reactor such as the Wuhrmann (Fig 5.1b) or Bardenpho (Fig 5.1c) configurations there is a net increase in the ammonia concentration in the anoxic reactor due to ammonification of TKN from lysed material less that due to deammonification for synthesis of heterotrophs. However, simulations as well as experimental results (Arkley and Marais, 1981) indicate that this increase is very small and can be neglected.

Accepting the approximations set out above, an expression for the effluent ammonia concentration in a series reactor system with pre- and/or post-denitrification is derived by performing a mass balance over the system on the nitrifying sludge mass, MX_n. Under steady state conditions the mass of nitrifiers does not change with time, i.e.

$$(dMX_n/dt) = 0 = (dMX_n/dt)_g + (dMX_n/dt)_d + (dMX_n/dt)_w \qquad (5.1)$$

where

MX_n = mass of nitrifiers in the system (mg VSS)

X_n = concentration of nitrifiers in the system (mg VSS.ℓ^{-1})

Indices g, d and w refer to growth, death and wastage respectively.

Let f_x = anoxic sludge mass fraction and $(1-f_x)$ = aerobic sludge mass fraction. Considering separately each term on the right hand side of Eq (5.1):

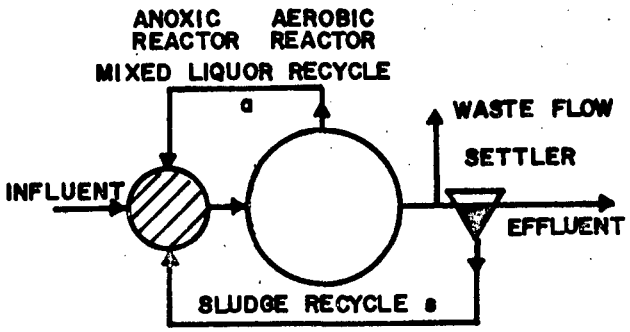

Fig. 5.1a *Modified Ludzack Ettinger configuration.*

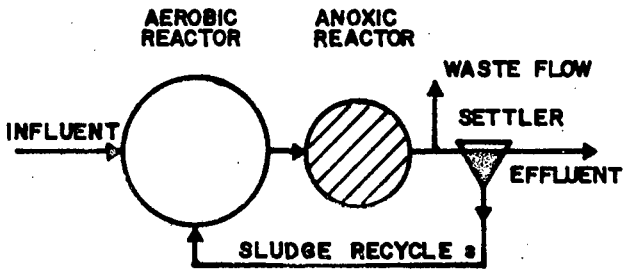

Fig. 5.1b *Wuhrmann configuration.*

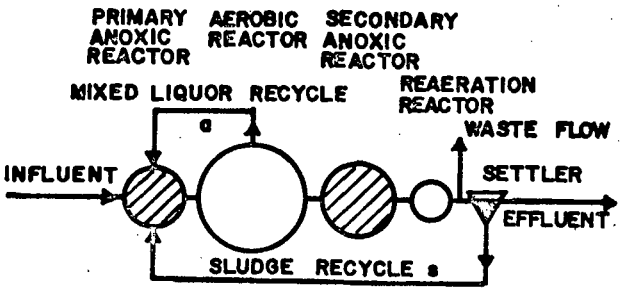

Fig. 5.1c *Bardenpho configuration.*

(1) Growth: This takes place only in the aerobic reactor and can be expressed in terms of the Monod Equation:

$$(dMX_n/dt)_g = V.(1-f_x)\ (dX_n/dt)_g$$

$$= V.(1-f_x)\ X_n.\mu_{nT}$$

$$= V(1-f_x)\ X_n\ \mu_{nmT}.N_a/(N_a+K_{nT}) \qquad (5.2)$$

where

μ_{nT}, μ_{nmT} = specific and maximum specific growth rate of nitrifiers respectively (d^{-1})

N_a = ammoniacal nitrogen concentration (mg NH_3-N.ℓ^{-1})

K_{nT} = half saturation value (mg NH_3-N.ℓ^{-1})

f_x = anoxic sludge mass fraction (mg VSS.mg VSS^{-1})

V = total reactor volume (ℓ)

(2) Death: This takes place in both aerobic and anoxic zones:

$$(dMX_n/dt)_d = V(dX_n/dt)_d = -Vb_{nT}X_n. \qquad (5.3)$$

where

b_{nT} = death rate constant for nitrifiers (mg X_n.mg $X_n^{-1}d^{-1}$)

(3) Sludge wastage: With a sludge waste flow $q = V/R_s$ (see Chapter 3, Section 3.9), the rate of wastage of nitrifiers is $q.X_n$ mg nitrifiers per day, i.e.,

$$(dMX_n/dt)_w = -q.X_n = -VX_n/R_s \qquad (5.4)$$

where

R_s = sludge age (d)

Substituting Eqs (5.2), (5.3) and (5.4) in Eq (5.1):

$$(dMX_n/dt) = 0 = (1-f_x)V.X_n.\mu_{nmT}.N_a/(N_a+K_{nT})-V.b_{nT}.X_n-V\,X_n/R_s \qquad (5.5)$$

Dividing by $V.X_n$ and solving for N_a:

$$N_a = K_{nT}(b_{nT}+1/R_s)/((1-f_x).\mu_{nmT}-(b_{nT}+1/R_s)) \qquad (5.6)$$

The effluent ammonia concentration as given by Eq (5.6), when compared with that predicted by the general model under constant flow and load conditions, shows insignificant differences irrespective of sludge age, reactor configuration, anoxic sludge fraction or temperature.

For completely aerobic systems, (f_x=0), Eq (5.6) reduces to the expression for the effluent ammonia concentration from a completely mixed aerobic reactor, developed in Chapter 3, Eq (3.217):

$$N_a = K_{nT}(b_{bT}+1/R_s)/(\mu_{nmT}-(b_{nT}+1/R_s)) \qquad (5.7)$$

Comparing Eqs (5.6) and (5.7) these differ only in that for an anoxic/aerobic system the maximum specific growth rate of nitrifiers μ_{nmT} is multiplied by the <u>aerobic</u> sludge mass fraction $(1-f_x)$ to give an apparent maximum specific growth rate, μ'_{nmT}, where

$$\mu'_{nmT} = \mu_{nmT}(1-f_x) \qquad (5.8)$$

i.e. the inclusion of unaerated reactors is equivalent to reducing the true maximum growth rate of nitrifiers, μ_{nmT}, to give an apparent maximum growth rate, μ'_{nmT}.

It is important to note that Eqs (5.6) and (5.7) presuppose that nitrification does take place, i.e. that the sludge age R_s is greater than the minimum sludge age for nitrification, R_{sm}. The minimum sludge age for nitrification in aerobic systems is defined as (Chapter 3, Eq (3.201a)):

$$R_{sm} = 1/(\mu_{nmT}-b_{nT}) \qquad (3.201a)$$

For anoxic/aerobic systems the minimum sludge age can be found by substituting μ'_{nmT} for μ_{nmT}:

$$R_{sm} = 1/(\mu'_{nmT}-b_{nT})$$

$$= 1/((1-f_m)\mu_{nmT} - b_{nT}) \qquad (5.9)$$

where

$(1-f_m)$ = aerobic sludge mass fraction below which no nitrification takes place.

2.2 Minimum aerobic sludge fraction for Nitrification

From Eq (5.9) it is evident that in anoxic/aerobic systems, for a particular temperature the minimum sludge age for nitrification depends on both the maximum specific growth rate of nitrifiers, μ_{nmT}, and the aerobic sludge mass fraction, $(1-f_x)$, or, from another point of view, at any specified sludge age there is a minimum aerobic sludge mass fraction $(1-f_m)$ below which no nitrification will take place. This minimum aerobic sludge mass fraction can be found by solving for $(1-f_m)$ from Eq (5.9):

$$(1-f_m) = (b_{nT}+1/R_s)/\mu_{nmT} \qquad (5.10)$$

In practice, the aerobic sludge fraction must be larger than $(1-f_m)$ because the objective of anoxic/aerobic systems, inter alia, is to guarantee efficient nitrification. At R_s slightly greater than R_{sm}, although nitrification is possible, nitrification will not be efficient; for "efficient" nitrification the minimum aerobic sludge mass fraction can be calculated from Eq (5.6) by specifying the desired maximum ammonia effluent concentration N_{ad}. Solving for $(1-f_x)$ from Eq (5.6) for a specified ammonia effluent concentration, N_{ad}, and substituting $(1-f_M)$ for $(1-f_x)$ to signify the minimum aerobic sludge mass fraction required to produce the desired ammonia effluent concentration N_{ad}:

$$1 - f_M = (1+K_{nT}/N_{ad})(b_{nT}+1/R_s)/\mu_{nmT} \qquad (5.11)$$

Eq (5.11) leads to the concept of a safety factor for efficient nitrification. Comparing Eqs (5.10) and (5.11) the minimum aerobic sludge mass fraction, $1-f_M$, required for nitrification to a desired effluent ammonia concentration, N_{ad}, is greater by a factor $(1+K_{nT}/N_{ad})$ than the aerobic sludge mass fraction below which no nitrification can take place, $(1-f_m)$. This factor can be termed the safety factor, S_f, i.e.

$$S_f = (1+K_{nT}/N_{ad}). \qquad (5.12)$$

The value of the safety factor depends on the desired effluent ammonia concentration, for example at 14°C for $K_{nT} = 0,5$ mg N.ℓ^{-1} and $N_{ad} = 2$ mg N.ℓ^{-1} the value of the safety factor is $S_f = 1+0,5/2 = 1,25$. The concept of a safety factor is most useful for practical design application, for another reason: The effluent ammonia concentration N_a is very insensitive to the sludge age once the sludge age exceeds the minimum for nitrification by about 25 per cent or more. This, in reverse, makes the desired sludge age very sensitive to the value chosen for a particular desired effluent ammonia concentration, N_{ad}, and may result in widely different sludge ages for the same process by choosing slightly different N_{ad} values. It is preferable, therefore, in design practice to choose a factor of safety and calculate the expected effluent ammonia concentration. That is, normally one would calculate $(1-f_m)$ from Eq (5.10) and then accept $(1-f_M) = S_f(1-f_m)$. From simulation using the general model, taking account of Monod kinetics for nitrification and cyclic load conditions (see Section 6 of this chapter) the safety factor should never ne less than 1,25 and preferably 1,5 or greater to ensure 95 per cent nitrification efficiency.

2.3 Maximum anoxic sludge mass fraction.

2.3.1 Requirements for efficient nitrification

In the section above it was shown that under constant flow and load conditions the minimum aerobic sludge mass fraction for efficient nitrification, $(1-f_M)$, must be at least 25 per cent greater than the aerobic sludge mass fraction below which no nitrification can take place, $(1-f_m)$. For a particular minimum aerobic sludge mass fraction, $1-f_M$, there is a corresponding maximum anoxic sludge mass fraction, f_M, which is given by rearranging Eq(5.12):

$$f_M = 1-(1+K_{nT}/N_{ad}).(b_{nT}+1/R_s)/\mu_{nmT} \qquad (5.13a)$$

$$= 1-S_f.(b_{nT}+1/R_s)/\mu_{nmT} \qquad (5.13b)$$

From the derivation of Eq(5.13a) the maximum anoxic sludge mass fraction, f_M, that does not inhibit efficient nitrification depends upon four factors:

(1) the desired ammonia effluent concentration, N_{ad}.

(2) the kinetic constants of nitrification (μ_{nmT}, b_{nT} and K_{nT}).

(3) the temperature of the mixed liquor.

(4) the operational sludge age R_s. i.e., the sludge age of the anoxic/aerobic system.

These factors refer only to the kinetic behaviour to ensure efficient nitrification and do not include constraints arising from other process requirements; these requirements are conveniently introduced after first considering the interplay of the factors listed above. The interplay of these factors can be illustrated by comparing the behaviour of the process for a specified effluent ammonia concentration N_{ad} at two temperatures, say 20°C and 14°C respectively. Select N_{ad} = 2 mg NH_3-N.ℓ^{-1} and plot f_M versus the operational sludge age R_s for different μ_{nmT} values.

(The relationship between the μ_{nmT} values at the two temperatures is given in Table 3.7: $\mu_{nm14} = \mu_{nmT20}\ (1,123)^{14-20}$). These plots are shown in Figs 5.2 (a & b) for 14°C and 20°C respectively. For any sludge age and a pair of μ_{nmT} values (for example $\mu_{nm20} = 0,4\ d^{-1}$ and $\mu_{nm14} = 0,4\ (1,123)^{14-20} = 0,2d^{-1}$) the maximum anoxic sludge mass fraction f_M at the lower temperature is always smaller than that at the higher temperature. Hence for design, the lower temperature will be the critical one insofar as the calculation of the maximum anoxic sludge mass fraction is concerned.

Also in Eq(5.12) (defining the safety factor S_f) by substituting the selected value of N_{ad} and the appropriate K_{nT} value ($K_{nT} = 1,0(1,123)^{T-20}$) the value of S_f will always be smaller at 14°C than at 20°C, in this case 1,25 and 1,5 respectively.

Basing the design on ammonia effluent concentration, as done above, one notes that the factor of safety is a function of temperature. Previously it was stated that it is preferable to base the design on a selected safety factor rather than accept an ammonia effluent quality as this leads to a more stable design, i.e. it is preferable to use Eq(5.13b) to Eq(5.13a). In this event,from what has been discussed above, in calculating f_M the selected value of S_f also must be applied to the lowest temperature. Furthermore, from Eq(5.13b) the magnitude of the minimum aerobic sludge mass fraction $(1-f_M)$ is inversely proportional to μ_{nmT} so that in calculating the maximum f_M, the smallest expected μ_{nmT} value should be used; again this will occur at the lower temperature.

The magnitude of μ_{nmT} is a sewage characteristic of the influent waste flow (see Chapter 4, Section 4) and can differ widely between different waste flows. Furthermore, μ_{nmT} can be determined only by experiment; as μ_{nmT} is of such vital importance in the process design of nitrification-denitrification systems one can appreciate the need for simple experimental techniques whereby μ_{nmT} can be evaluated quickly and reliably.

To summarize, the design procedure for nitrification in nitrification-denitrification systems requires inter alia calculation

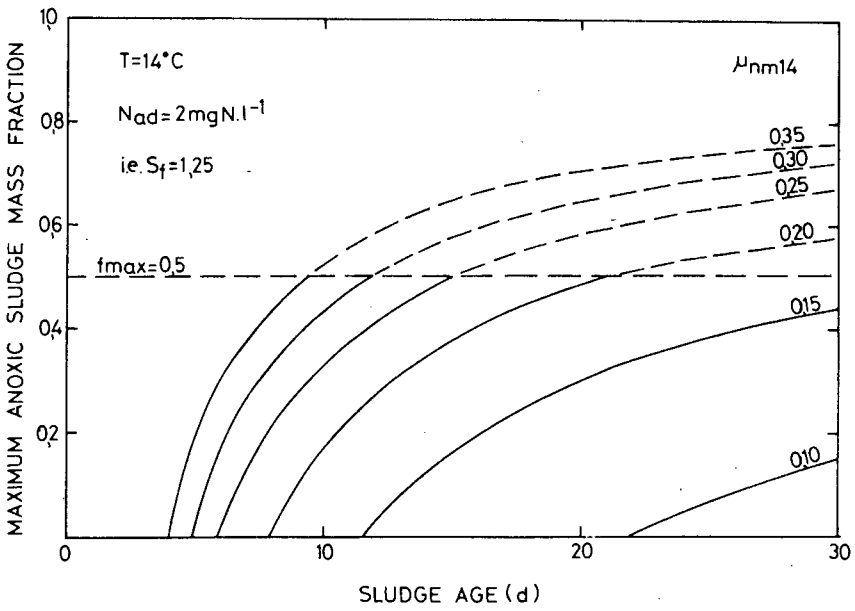

Fig. 5.2a *Maximum anoxic sludge mass fraction for an effluent ammonia concentration N_{ad} = 2 mg l^{-1} (i.e. S_f = 1,25) as a function of the sludge age for a temperature T = 14°C and different μ_{nm14} values.*

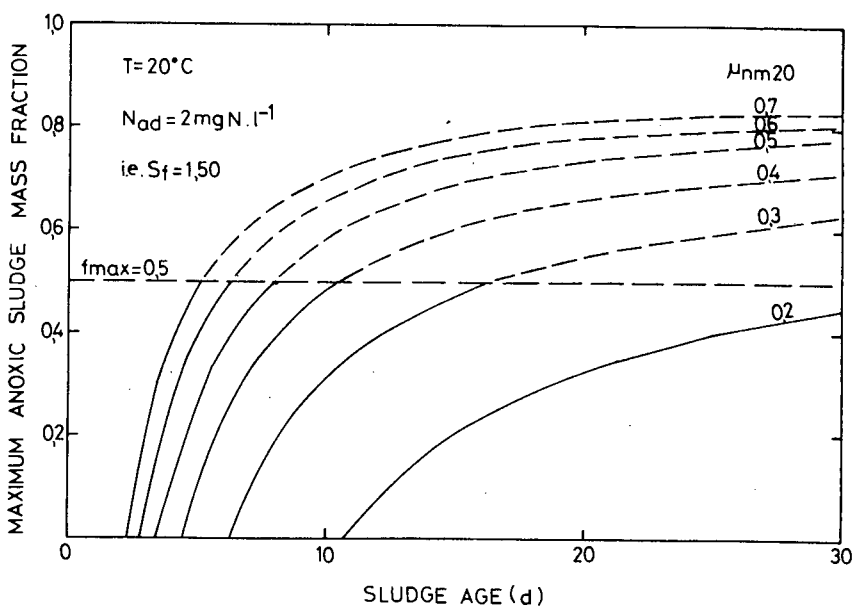

Fig. 5.2b *Maximum anoxic sludge mass fraction f_M for an effluent ammonia concentration N_{ad} = 2 mg N/l (i.e. S_f = 1,5) as a fraction of the sludge age, R_s, for a temperature T = 20°C and different μ_{nm20} values.*

of the allowable maximum anoxic sludge mass fraction based on (1) a selected factor of safety (S_f); (2) the operational sludge age; (3) the minimum temperature expected in the plant and (4) the maximum specific growth rate of nitrifiers at that temperature. If the procedure to determine the maximum anoxic sludge mass fraction is followed for the lowest temperature expected in the plant, then efficient nitrification at any temperature higher than the minimum will always be guaranteed.

2.3.2 Settling properties of sludge

The discussion above refers only to the kinetic behaviour for efficient nitrification. Another factor, not in this category, must now be considered. In Figs 5.2 (a and b) theoretically it is possible to have a very high anoxic sludge mass fraction if the μ_{nmT} value in a particular waste is high. However, from experimentally observed behaviour (Arkley and Marais, 1981) it would appear that the anoxic sludge mass fraction should not exceed about 50 per cent, principally because the settling properties of the sludge deteriorate sharply when the anoxic sludge fraction increases above $f_{max} = 0,5$.

In Figs 5.2 (a and b) the upper limit due to this new constraint ($f_M = f_{max}$) is drawn in and imposes a restriction on the design of the plant. Hence in the design of anoxic/aerobic activated sludge systems there are two constraints that may limit the maximum anoxic sludge mass fraction, i.e.

(1) the requirement for efficient nitrification (Eq (5.13b))

(2) the requirement that the anoxic sludge mass fraction must not exceed 50 per cent of the total sludge mass.

2.4 Nitrification potential and nitrification capacity

The nitrification potential, N_p, is defined as the concentration of influent TKN that is available for nitrification. In Chapter 3,

Section 9.4, it was pointed out that only a fraction of the influent TKN, N_{ti}, is actually available for nitrification. A certain concentration N_s is required for incorporation in the wasted sludge and in addition there is always a residual organic nitrogen concentration, N_o, in the effluent. Hence the nitrification potential, N_p, is given by:

$$N_p = N_{ti} - N_s - N_o \tag{5.14}$$

The nitrification capacity (Chapter 3, Section 9.4) is defined as the concentration of influent TKN in a nitrification-denitrification system that is converted to nitrate. Taking into account the effluent ammonia concentration but disregarding the concentration of nitrite, the nitrification capacity is given by the difference between the ammonia concentration available for nitrification (i.e. the nitrification potential) and the effluent ammonia concentration.

$$N_c = N_p - N_a$$

$$= N_{ti} - N_s - N_o - N_a \tag{5.15}$$

In Eq(5.15) the values of N_s and N_o for a particular design situation are known Eqs(3.189 and 3.195 respectively), i.e.

$$N_s = f_n(Y_h(1+f\ b_{hT}R_s)\ (1-f_{us}-P.f_{up})/(1+b_{hT}R_s)+f_{up})S_{ti} \tag{5.16}$$

$$N_o = [N_{oi}+f_n(1-f)b_{hT}X_a/(1-P\ Y_h)\ ]\ /(1+K_{rT}X_aR_h) \tag{5.17}$$

Furthermore, the value of N_a is determined from Eq(5.6). Consequently for any influent TKN value under constant flow and load conditions the nitrification potential and nitrification capacity can be calculated. From Eq(5.6) it is clear that the nitrification capacity depends upon the anoxic sludge mass fraction. This is illustrated in Fig 5.3 where the nitrification capacity and the nitrification potential are shown plotted as a function of the anoxic sludge mass fraction, using the following input data and sewage characteristics:

$$T = 14^{o}C;\ \ R_s = 20d;\ \ S_{ti} = 500\ \text{mg COD}.\ell^{-1};\ \ N_{ti} = 50\ \text{mg N}.\ell^{-1}$$

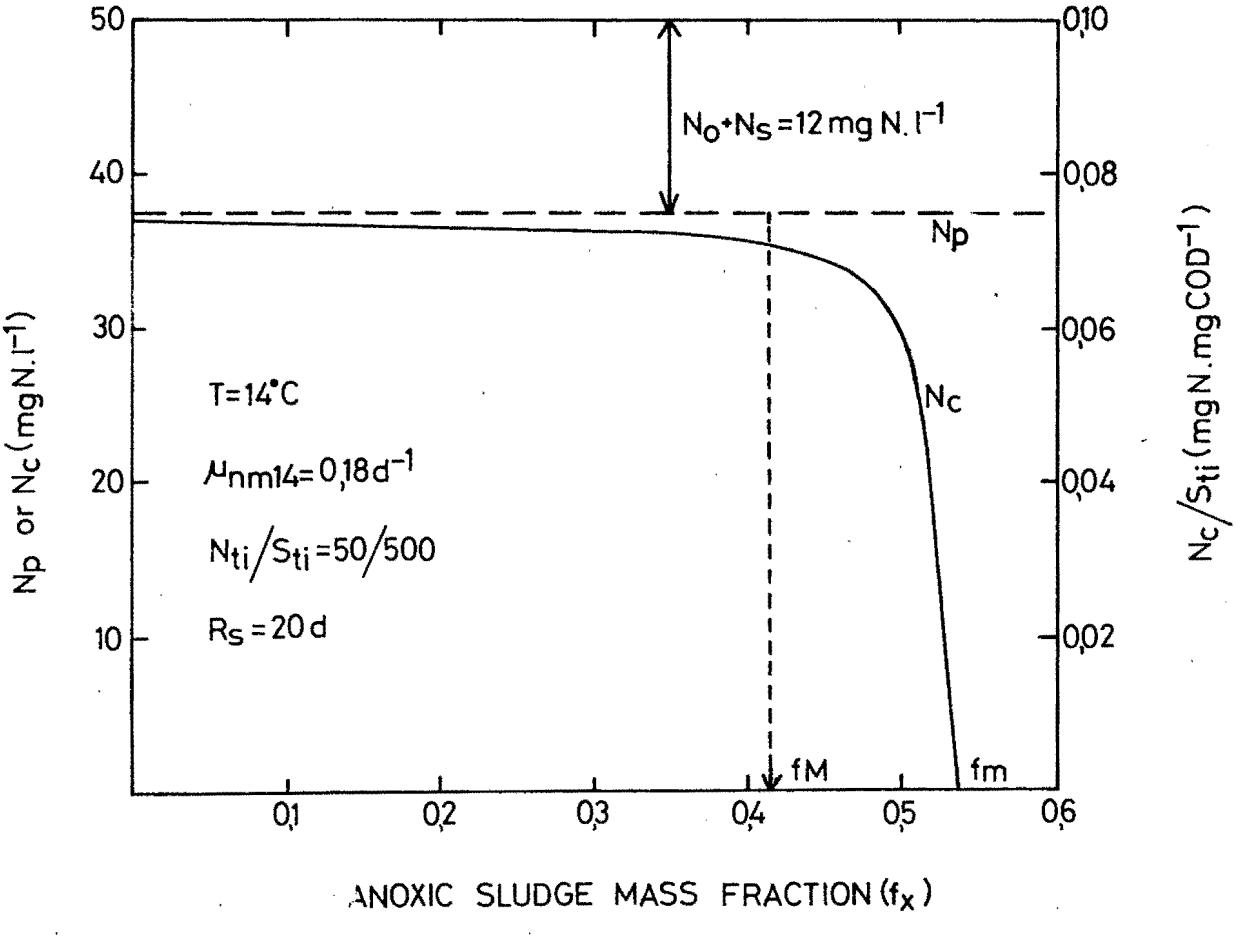

Fig. 5.3 *Nitrification capacity, N_c, as a function of the anoxic sludge fraction, f_x, for R_s = 20 d T = 14°C, μ_{nm14} = 0,18 d^{-1} and N_{ti}/S_{ti} = 50/500.*

μ_{nm14} = 0,18 d^{-1}; N_o = 1,0 mg N.ℓ^{-1}; f_{us} = 0,1 and f_{up} = 0,09 mg VSS.mg COD^{-1}, i.e. from Eq(5.16): $N_s = 0{,}022.S_{ti}$ = 11 mg N.ℓ^{-1}.

Hence $N_p = N_{ti} - N_s - N_o = 50 - 11 - 1 = 38$ mg N.ℓ^{-1}.

The values of $f_m = 1 - (b_{nT}+1/R_s)/\mu_{nm14} = 0{,}53$ and of $f_M = 1 - S_f (b_{nT}+/R_s)/\mu_{nm14} = 0{,}42$ (S_f = 1,25) are also indicated in Fig. 5.3

Normally efficient nitrification is a basic requirement in design of nitrification-denitrification systems. Also, later it will be shown that one wishes to remove all the nitrate by denitrification, or, where this is not possible, to remove the maximum amount without impairing the efficiency of nitrification. In order to do the combined analysis of nitrification and denitrification, it is necessary inter alia to evaluate the nitrification capacity subject to the condition that at any sludge age denitrification must be maximum, i.e., that the anoxic sludge mass fraction must have the maximum allowable value, f_M. The minimum sludge age allowing efficient nitrification is given by:

$$R'_{sm} = S_f R_{sm} \qquad (5.17a)$$

where

$$R_{sm} = 1/(\mu_{nmT} - b_{nT})$$

Knowing the maximum anoxic sludge mass fraction as a function of the sludge age (using Eq 5.13b) the corresponding nitrification capacity, N_c, can be determined at any sludge age $R_s > R'_{sM}$ using (Eq 5.15):

$N_c = N_{ti} - N_s - N_o - N_a$. The value of the influent TKN, N_{ti}, is measured on the influent flow; N_s, N_o and N_a are calculated from Eqs(5.16, 5.17 and 5.6) respectively. In order to calculate N_s and N_o at the different sludge ages it is necessary to know the influent COD concentration, S_{ti}, and the characteristics f_{us}, f_{up} and f_{na}. In Fig 5.4, N_c is shown plotted as a function of sludge age for a series of N_{ti}/S_{ti} ratios with S_{ti} = 500 mg COD.ℓ^{-1}; T = 14°C; f_{us} = 0,10; f_{up} = 0,09 mg VSS.mg COD^{-1} and f_{na} = 0,75.

To calculate the effluent ammonia concentration, a safety factor S_f = 1,25 has been assumed (i.e. N_a = 2 mg N.ℓ^{-1}). The maximum anoxic sludge mass fraction for the assumed value of μ_{nm14} = 0,18 d^{-1} is also plotted in Fig 5.4

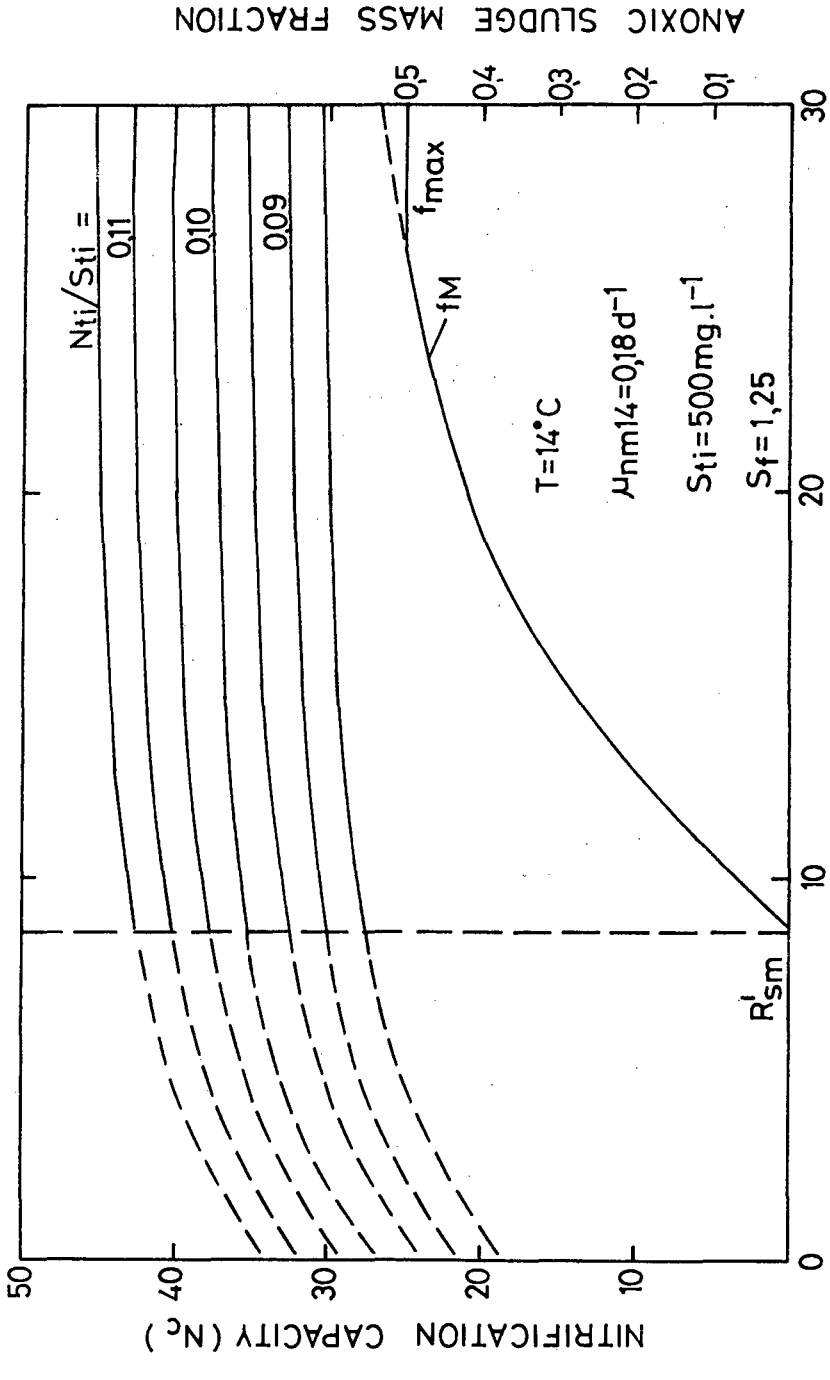

Fig. 5.4 *Nitrification capacity as a function of the sludge age and maximum anoxic sludge fraction f_M as a function of the sludge age.*

3. DENITRIFICATION POTENTIAL

The denitrification potential of a system, i.e. the maximum system nitrate removal possible in a specific configuration has already been discussed in Chapter 2, Section 3.7. In pre-denitrification systems the denitrification potential was shown to be linked to the utilization of easily and of slowly biodegradable material (Chapter 4, Section 2).

$$D_{p1} = \Delta N'_{1s} + \Delta N''_{1s}$$

$$= (\alpha + K_2 C_r f_{x1}) S_{bi} \qquad (5.18)$$

where

D_{p1} = denitrification potential of a predenitrification reactor with an anoxic sludge mass fraction f_{x1}

$\Delta N'_{1s}$ = maximum system nitrate removal due to utilization of easily biodegradable material.

$$= \alpha \cdot S_{bi}$$

$$S_{bi} = (1-f_{us}-P.f_{up}) S_{ti}$$

$$\alpha = f_{ca} (1-PY_h)/2,86$$

f_{ca} = ratio of easily biodegradable:total biodegradable influent COD.

$$= S_{bsi}/S_{bi}$$

$\Delta N''_{1s}$ = maximum system nitrate removal due to utilization of slowly biodegradable material

$$= K_2 X_a R_1$$

$$= K_2 C_r f_{x1} S_{bi}$$

K_2 = denitrification rate constant

C_r = sludge mass factor

$$= Y_h R_s/(1+b_{hT}R_s) \qquad (5.18a)$$

Equation(5.18) is only valid if the anoxic sludge fraction is sufficiently large to allow complete utilization of easily biodegradable material

i.e. $f_{x1} > f_{min} = \alpha/(K_1 C_r)$ (See (Eq 2.76))

For $f_{x1} < f_{min}$ the denitrification potential is given by:

$$D_{p1} = (K_1+K_2)C_r f_{x1} S_{bi} \qquad (5.18b)$$

In a post-denitrification system denitrification is due to utilization of slowly biodegradable material only. The denitrification potential of a postdenitrification reactor is given by:

$$D_{p3} = K_3 C_r f_{x3} S_{bi} \qquad (5.19)$$

In a system with both a pre- and post-denitrification reactors the total denitrification potential is:

$$D_p = D_{p1} + D_{p3}$$

$$D_p = (\alpha + K_2 C_r f_{x1} + K_3 C_r f_{x3}) S_{bi} \quad (f_{x1} > f_{min}) \text{ or} \qquad (5.20)$$

$$D_p = ((K_1+K_2)C_r f_{x1} + K_3 C_r f_{x3}) S_{bi} \quad (f_{x1} < f_{min}) \qquad (5.20a)$$

The values of the constants K_1, K_2, K_3 and α have been determined in Chapter 2 from experimental data by Marais and his coworkers,

$$K_1 = 0{,}03(1{,}20)^{T-20} \quad \text{mg } NO_3\text{-N.mg } X_a^{-1}.h^{-1} \qquad (5.21)$$

$$K_2 = 0{,}0042(1{,}08)^{T-20} \quad \text{mg } NO_3\text{-N.mg } X_a^{-1},h^{-1} \qquad (5.22)$$

$$K_3 = 0{,}0032(1{,}03)^{T-20} \quad \text{mg } NO_3\text{-N.mg } X_a^{-1}.h^{-1} \qquad (5.23)$$

$$\alpha = 0{,}028 \text{ mg } NO_3\text{-N.mg } S_{bi}^{-1} \qquad (5.24)$$

In Chapter 4 it was shown that the values of the denitrification rate constants (K_1, K_2, K_3) and α depend on the sewage characteristics and ideally must be determined experimentally for each waste flow. However, the values defined by Eqs(5.21 to 5.24) probably are representative of the bulk of municipal waste flows, i.e. flows containing a large fraction of domestic sewage and a minor fraction of industrial waste and in the following sections these values will be used in numerical examples.

From Eq(5.20) the denitrification potential is proportional to the influent biodegradable COD concentration and, furthermore, depends upon the anoxic sludge mass fraction, f_{x1} and f_{x3}, the sludge age, R_s, and the temperature, T, of the system. In Fig 5.5 are plotted the denitrification potentials of a pre- and a post-denitrification system respectively, as a function of the anoxic sludge mass fraction (lines A and B respectively). The following assumptions have been made, T = 14°C R_s = 20 d; S_{ti} = 500 mg COD/1; S_{bi}= 0,8.S_{ti} ;f_{ca} = 0,24 mg COD.mg COD.$^{-1}$.

Figure 5.5 highlights the considerable difference that exists between the denitrificaton potentials of equal sized pre- and post-denitrification reactors in a Modified Ludzack Ettinger and in a Wuhrmann system respectively. Note that the lines A and B at 14°C are parallel for $f_{x1} > f_{min}$, because at this temperature $K_2 = K_3$. At temperatures T >14°C, always $K_2 > K_3$ and the difference in denitrification potential between pre- and post-denitrification reactors of equal size tends to increase as the anoxic sludge mass fraction increases. This is illustrated in Fig 5.5 (dotted lines) for 20°C.

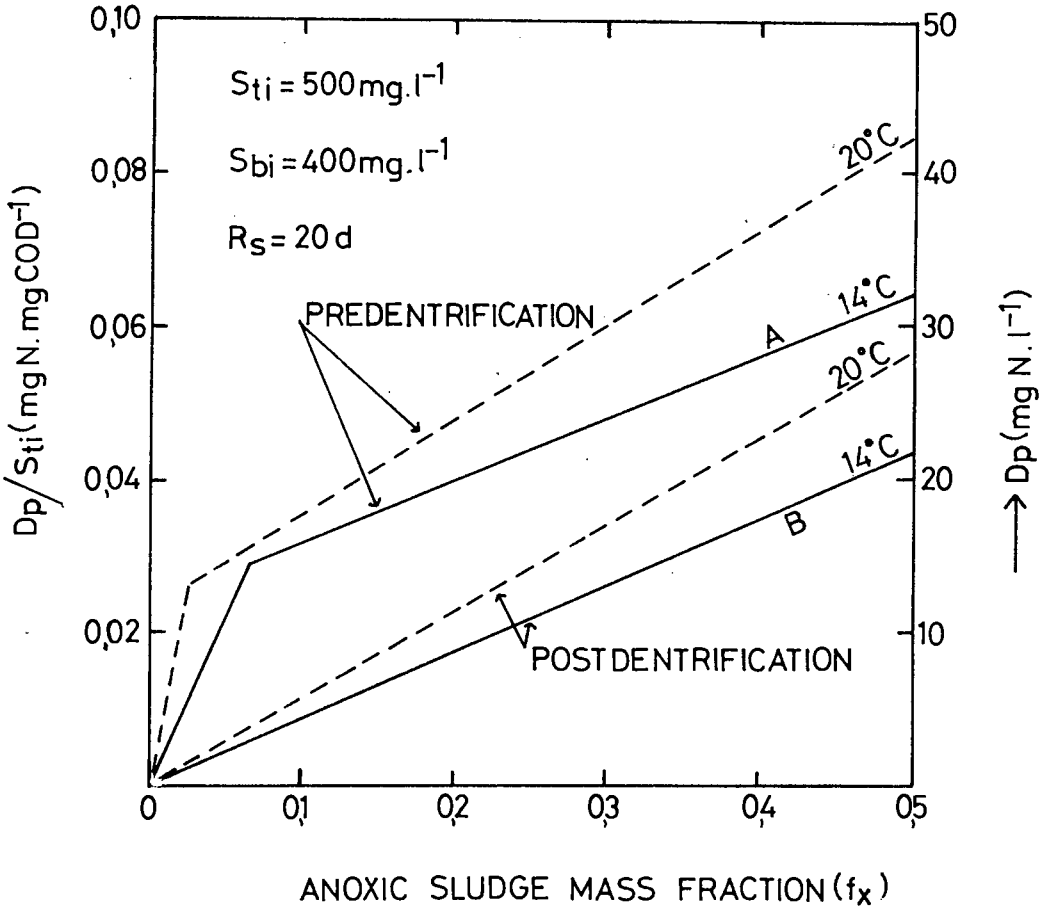

Fig. 5.5 *Denitrification potential as a function of the anoxic sludge mass fraction for a pre and post-denitrification reactor at 14°C (A and B respectively) and at 20°C (interrupted curves).*

In practical nitrification-denitrification systems the denitrification potential is limited by the maximum allowable value of the anoxic sludge fraction f_M. For a pre-denitrification system:

$$D_{p1} = (\alpha + K_2 C_r f_M) S_{bi} \quad (f_{x1} = f_M) \tag{5.25}$$

For a post-denitrification system:

$$D_{p3} = K_3 C_r f_M S_{bi} \quad (f_{x3} = f_M) \tag{5.26}$$

and for a system with both pre- and post-denitrification:

$$D_p = (\alpha + K_2 C_r f_{x1} + K_3 C_r f_{x3}) S_{bi} \quad (f_{x1} + f_{x3} = f_M) \tag{5.27}$$

where C_r is defined by (Eq 5.18a)

Both the maximum anoxic sludge mass fraction, f_M, (Eq(5.13b)) and the sludge mass factor, C_r, (Eq(5.18a)) are functions of the sludge age. Consequently the denitrification potentials Eq(5.25 to 5.27) in pre- or post-denitrification reactors are also functions of the sludge age. The maximum anoxic sludge mass fraction, f_M, is subject to an upper limit: $f_M = f_{max} = 0,5$. The sludge age where f_M attains the value of f_{max} is designated R_{so}. Using Eqs(5.25 and 5.26) plots of the denitrification potential versus sludge age are shown in Fig 5.6 for a pre- and post-denitrification system respectively using T = 14°C; S_{ti} = 500 mg COD.ℓ^{-1}; and S_{bi} = 400 mg.ℓ^{-1}. In order to calculate f_M in Eqs(5.25 and 5.26) it was assumed that $\mu_{nm14} = 0,18 d^{-1}$ and S_f = 1,25 (f_{max} = 0,5).

From Fig 5.6 there is an increase of the denitrification potential with increasing sludge age both for a pre- and a post-denitrification system. The increase is not continuous: In the range $R'_{sm} < R_s < R_{so}$ (corresponding to $0 < f_M < f_{max}$) the magnitude of the denitrification potentials increases rapidly. This is because in this region an increase of sludge age results in increases of both f_M and C_r in Eqs (5.25 and 5.26). In contrast in the region of sludge ages $R_s > R_{so}$ there is only a slow increase of the denitrification potential with increasing sludge age. In this region the value of C_r only increases, the anoxic

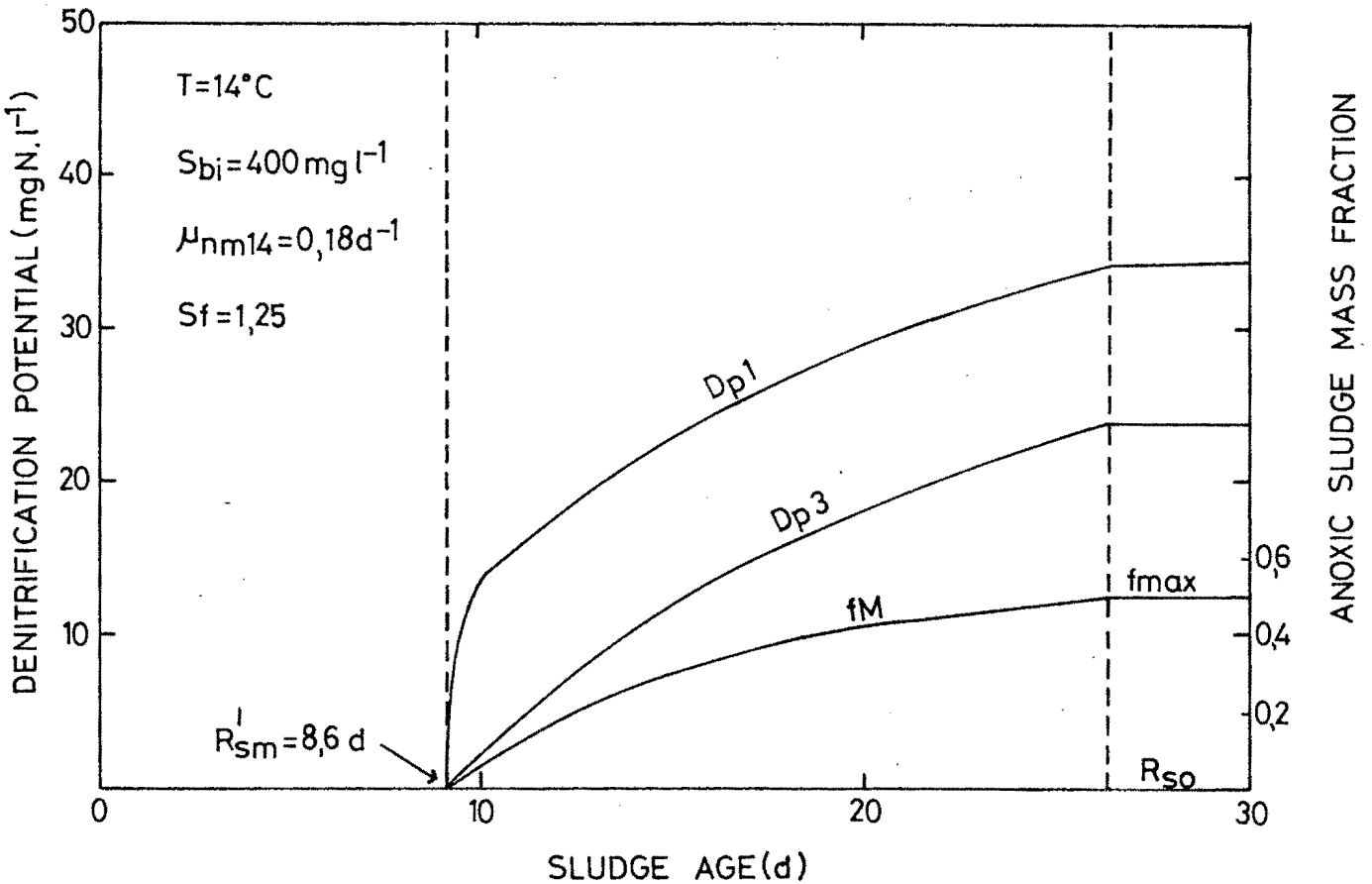

Fig. 5.6 *Denitrification potential in a pre-denitrification reactor (D_{p1}) and in a post-denitrification reactor(D_{p3}) and maximum anoxic sludge mass fraction (f_M) as a function of sludge age.*

sludge mass fraction remains fixed at $f_M = f_{max} = 0,5$.

3.1 Influence of dissolved oxygen

In single sludge nitrification-denitrification systems the influent flow and the recycle flows from the aerobic reactor and from the settler all may contain dissolved oxygen which is conveyed to the anoxic reactors. Normally dissolved oxygen is not detected in anoxic reactors as any oxygen introduced is utilized preferentially by the microorganism for electron transfer. If the assumption is made that the introduction of a small mass flow of dissolved oxygen into an anoxic reactor does not affect the rate of anoxic substrate utilization, the reduction of dissolved oxygen implies that the extent of reduction of nitrate will decrease correspondingly. Hence a denitrification capacity, D_c, could be defined as the difference between the denitrification potential, D_p, and the system oxygen removal in anoxic reactors, D_o, (expressed as mg equivalent nitrate per liter influent):

$$D_c = D_p - D_o \tag{5.28}$$

where

D_c = Denitrification capacity

D_p = Denitrification potential

D_o = System equivalent nitrate removal.

The value of D_o can be estimated from the DO concentration in the combined influent to an anoxic reactor: For a pre-denitrification reactor:

$$D_{o1} = (DO_i + a\ DO_a + sDO_s)/2,86 \tag{5.28a}$$

For a post-denitrification reactor

$$D_{o3} = (s+1)\ DO_s/2,86 \tag{5.28b}$$

where

D_{o1} = system equivalent nitrate removal in a pre-denitrification reactor (mg N.ℓ^{-1})

D_{o3} = system equivalent nitrate removal in a post-denitrification reactor (mg N.ℓ^{-1})

DO_i = DO concentration in the influent (mg O.ℓ^{-1})

DO_a = DO concentration in the 'a' recycle flow (mg O.ℓ^{-1})

DO_3 = DO concentration in the underflow (mg O.ℓ^{-1})

In practical nitrification-denitrification systems ($f_x > 0,3$) using realistic values for DO concentrations (< 2 mg O.ℓ^{-1}) and recycle ratios ($a+s < 5$) the influence of the DO on the extent of denitrification that can be attained is usually very small. A numerical example will illustrate this point: Assume a Bardenpho configuration operating at a sludge age of 20 d and at 14°C, treating sewage with a biodegradable influent concentration $S_{bi} = 400$ mg.ℓ^{-1} ($f_{ca} = 0,24$), having anoxic sludge mass fractions $f_{x1} = f_{x3} = 0,25$ for the pre- and post-denitrification reactor respectively, recycle ratios $a = 4$ and $s = 1$ and a dissolved oxygen concentrations of 1,4 mg O.ℓ^{-1} in the aerobic reactor and the settler. The denitrification potential of the system is (Eq(5.27)): $D_p = 35$ mg N.ℓ^{-1}. The system oxygen removals in the anoxic reactors (as equivalent nitrate) are $D_{o1} = 2,5$ and $D_{o3} = 1,0$ mg N.ℓ^{-1} in the pre- and post-denitrification reactor respectively (Eqs(5.28 and 5.29)). Thus, although there are two anoxic reactors and the recycle ratios are high the maximum possible reduction of the extent of nitrate removal will not exceed 10 per cent of the denitrification potential.

For the purpose of maximization of nitrogen removal in single sludge nitrification-denitrification systems, i.e., optimization of the reactor configuration, reactor volumes and recycle ratios, it is possible to include the influence of dissolved oxygen in the influent and in the recycle flows. The solutions obtained, when compared with optimal solutions neglecting the effect of dissolved oxygen, differ only marginally. The calculations however, are more complex by an order

of magnitude, compared to those when the effect of dissolved oxygen is neglected. If one considers that the optimization technique in any case does not take into account variations of the daily cyclic flow and load pattern of the influent nor random fluctuations from day to day in kinetic constants, the optimal solution neglecting the effect of dissolved oxygen in the recycles is completely adequate for design. Consequently, in the following sections the determination of optimal systems for maximum nitrogen removal will be discussed assuming that the dissolved oxygen concentration in the recycles is zero.

4. APPLICATION OF NITRIFICATION CAPACITY AND DENITRIFICATION POTENTIAL

In anoxic/aerobic activated sludge systems under constant flow and load conditions, the extent of nitrification is given by the nitrification capacity.* Provided nitrate is always available, the extent of denitrification is given by the denitrification potential. The application of these concepts to the calculation of the extent of nitrogen removal is exemplified in the analysis of a Modified Ludzack-Ettinger configuration (Fig 5.1a). The nitrification capacity, N_c, and the denitrification potential, D_{p1}, of a Modified Ludzack Ettinger configuration can be calculated as a function of the anoxic sludge mass fraction, f_{x1}, using the data from the previous plots, i.e. $T = 14^oC$; $R_s = 20$ d; $\mu_{nm14} = 0{,}18\ d^{-1}$; $N_{ti}/S_{ti} = 50/500$ mg N.mg COD^{-1}; $S_{bi}/S_{ti} = 0{,}8$; $N_o = 1$ mg N.ℓ^{-1}; $N_s = 11$ mg N.ℓ^{-1} and $a = 3$; $s = 1$. For these conditions the nitrification capacity and the denitrification are calculated from (Eqs 5.15 and 5.25) respectively. For the nitrification capacity:

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* The nitrification potential is given by the influent TKN available for nitrification, but as some ammonia is not nitrified, the nitrification capacity is always smaller than the potential.

$$N_c = N_{ti} - N_s - N_o - N_a$$

$$= N_p - N_a$$

$$= 50 - 11 - 1 - N_a$$

$$= 38 - 0,042/(0,18(1-f_{x1}) - 0,084)$$

For the denitrification potential

$$D_{p1} = (\alpha + K_2 C_r f_{x1}) S_{bi}$$

$$= (0,028 + 0,117 f_{x1}) S_{bi}$$

In fig 5.8 the values of N_c and D_{p1} are shown plotted as a function of the anoxic sludge mass fraction, f_{x1}. Also shown plotted is the nitrification potential $N_p = N_{ti} - N_s - N_o = 38,0$ mg N.ℓ^{-1}

In a Modified Ludzack-Ettinger configuration only a fraction $(a + s)/(a + s + 1)$ of the nitrate generated in the aerobic reactor (i.e. of the nitrificaton capacity, N_c) is recycled to the anoxic reactor (Fig 5.1a). Hence the nitrate <u>available</u> for denitrification in the predenitrification reactor, N_{av1}, is given by:

$$N_{av1} = N_c.(a + s)/(a + s + 1) \qquad (5.29)$$

Accepting an 'a' recycle (Fig 5.1a) of a = 3 and an underflow recycle s = 1, N_{av1} is calculated from (Eq 5.29) and shown plotted in Fig 5.8 as a function of f_{x1}.

From Fig 5.8 the effluent concentrations for ammonia and nitrate can be determined graphically as a function of the anoxic sludge mass fraction: Taking $f_{x1} = 0,2$ as an example, the ammonia effluent concentration, N_a, is given by the difference between the nitrification potential, N_p, and the nitrification capacity, N_c. The effluent nitrate concentration, N_{ne}, is obtained by comparing the nitrate available for denitrication, N_{av1}, to the denitrification potential, D_{p1}. If $N_{av1} > D_{p1}$ the load of nitrate to the anoxic reactor

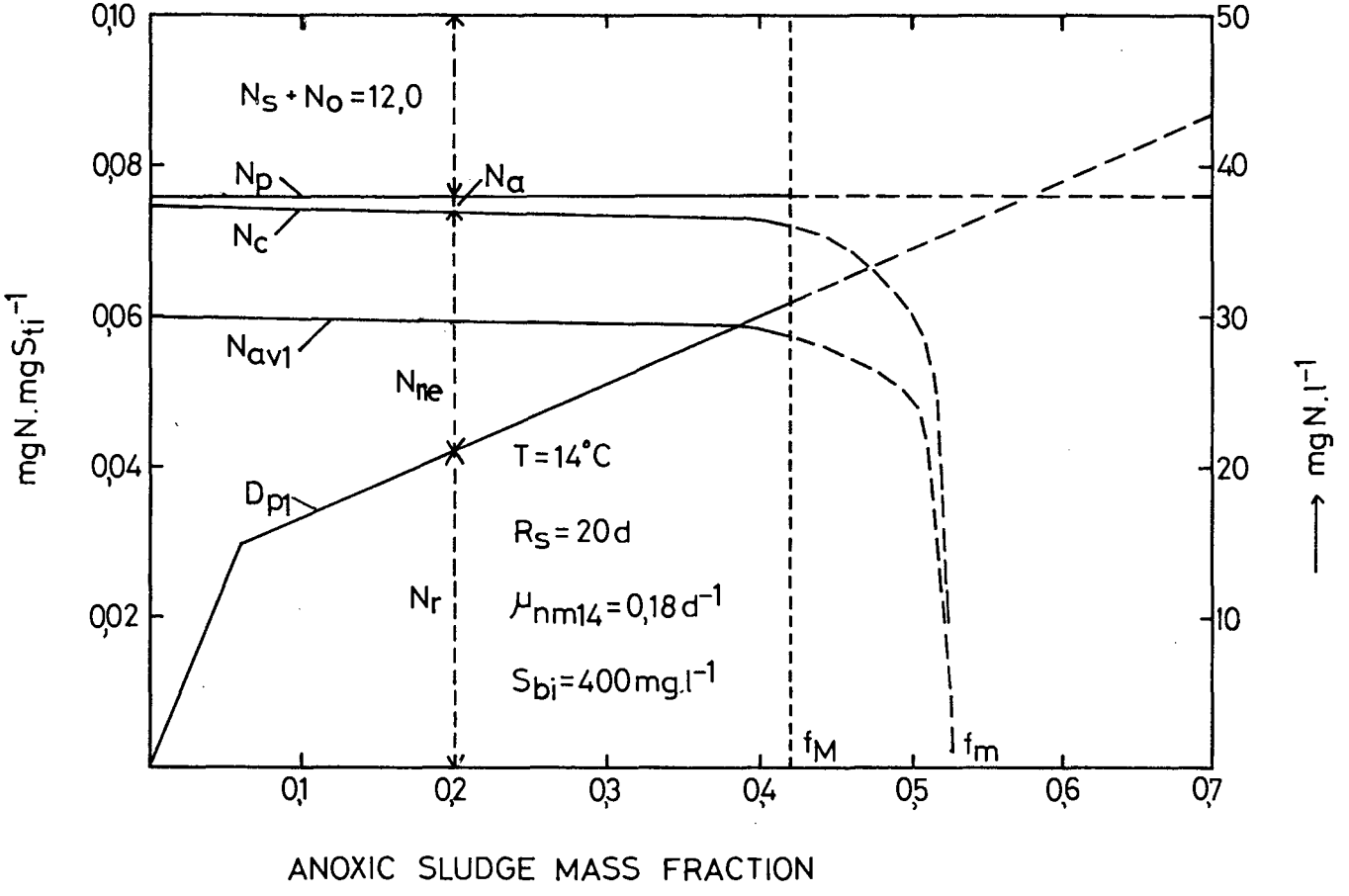

Fig. 5.8 *Nitrification potential (N_p), nitrification capacity (N_c), denitrification potential (D_p) and nitrate available for denitrification (N_{avl} for a=3 and s=1) as a function of the anoxic sludge mass fraction in a predenitrification system at 14°C.*

exceeds its potential for denitrification; in this event denitrification in the anoxic reactor is incomplete (i.e. nitrate is present in the effluent from the reactor) but the system nitrate removal, N_r, is the maximum that can be obtained for the particular anoxic sludge mass fraction, i.e. it is equal to D_{pl}. Hence the effluent nitrate concentration is given by the difference between the nitrification capacity and the denitrification potential, i.e.

$$N_{ne} = N_c - D_{pl} \qquad (N_{avl} > D_{pl}) \tag{5.30}$$

If $N_{avl} < D_{pl}$ denitrification is complete in the anoxic reactor and the reactor is in an "anaerobic" state: all the nitrate recycled to the anoxic reactor, i.e. N_{avl}, is removed. The reactor could remove more nitrate if the recycle ratios were increased. In this case the effluent nitrate concentration is given by the difference between the nitrification capacity and the nitrate available for denitrification, i.e.

$$N_{ne} = N_c - N_{avl} = N_c/(a + s + 1) \qquad (N_{avl} < D_{pl}) \tag{5.30a}$$

In Fig 5.9 the effluent ammonia and nitrate concentrations, N_a and N_{ne} respectively, obtained from Fig 5.8, are shown plotted as a function of the anoxic sludge mass fraction. Also shown is the total soluble effluent nitrogen concentration, $N_t = N_{ne} + N_a + N_o$. (N_o is assumed to have a constant value of 1 mg N.ℓ^{-1}). From Fig 5.9 the total soluble effluent nitrogen concentration is minimum when $N_{avl} = D_{pl}$ which in this example occurs at $f_{x1} = 0{,}38$. At this anoxic sludge mass fraction the value of N_t may be read off from Fig 5.9 or determined analytically:

$$N_a = K_{nT}\,(b_{nT} + 1/R_s) \;/\; [\,(\mu_{nmT}(1-f_{x1}) - (b_{nT} + 1/R_s)\,] = 1{,}5 \text{ mg N.}\ell^{-1}$$

$$N_{ne} = N_c - D_{pl} = N_c/(a+s+1) = 7{,}3 \text{ mg N.}\ell^{-1}$$

$$N_o = 1{,}0 \text{ mg N.}\ell^{-1}$$

i.e.

$$N_t = 1{,}5 + 7{,}3 + 1{,}0 = 9{,}8 \text{ mg N.}\ell^{-1}$$

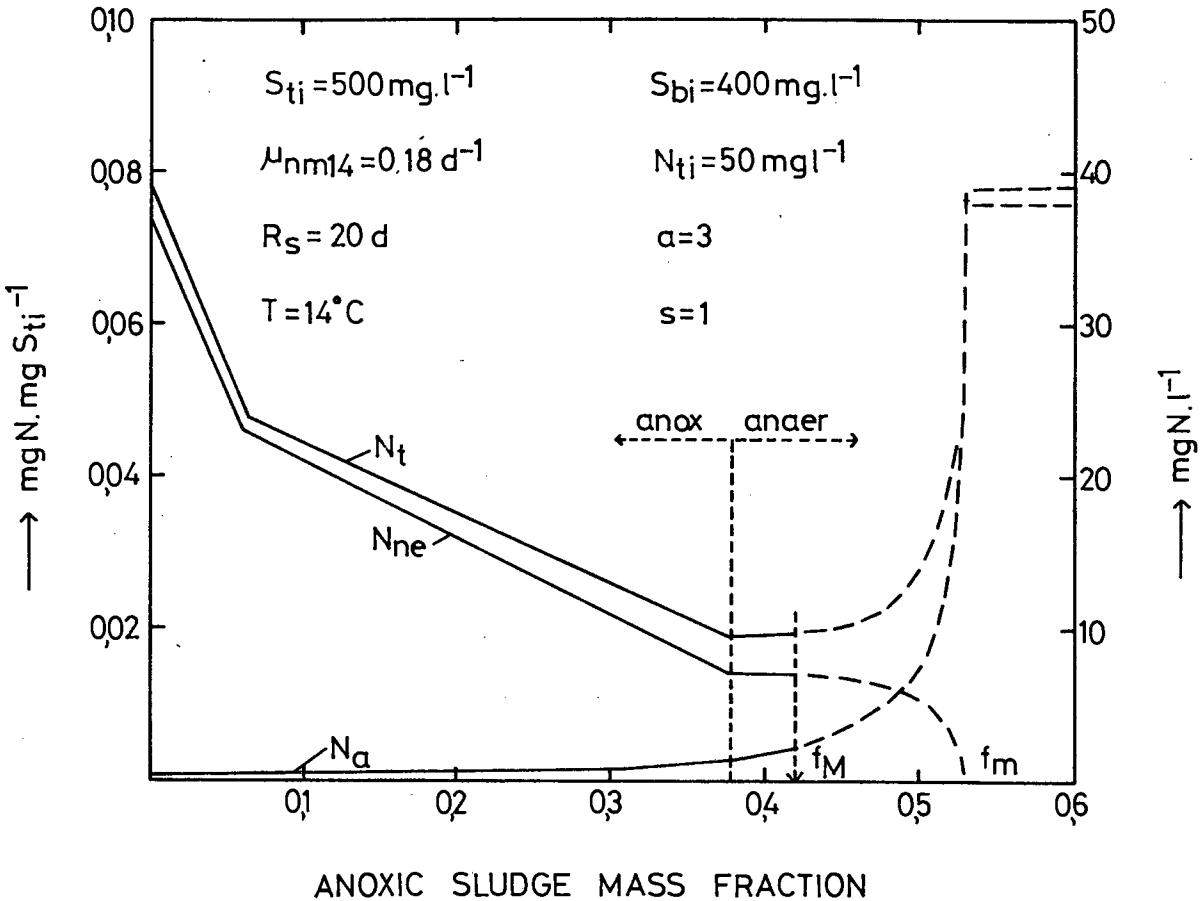

Fig. 5.9 *Concentration of nitrogenous species : nitrate(N_{ne}), ammonia (N_a) and total soluble nitrogen(N_t) in the effluent of a predenitrification system as a function of the anoxic sludge mass fraction (the effluent organic nitrogen concentration is estimated at N_o=1 mgM.$^{-1}$)*

The anoxic sludge mass fraction for minimum total soluble nitrogen in the effluent, $f_{x1} = 0,38$ in the above example is less than the maximum anoxic sludge mass fraction, f_M, for $S_f = 1,25$ (i.e. $N_a = 2$ mg $N.\ell^{-1}$); for the given conditions: $f_M = 0,42$ (Eq 5.13b). For $f_{x1} = f_M$ the denitrification potential $D_{pl} = (\alpha + K_2 C_r f_M)S_{bi} = (0,028 + 0,117.0,42)$ $400 = 30,9$ mg $N.\ell^{-1}$ and this value exceeds the nitrate available for denitrification N_{avl} $(N_p - N_a)(a + s)/(a + s + 1) = (3,80 - 2,0)$. $4/5 = 28,8$ mg $N.\ell^{-1}$. However, it is possible to increase the nitrate available for denitrification by increasing the 'a' and/or 's' recycle ratios. The required recycle ratio to balance the denitrification potential of the anoxic reactor with the nitrates available for denitrification can be determined from

$$D_{pl} = N_{avl} = (a+s)/(a+s+1)\ N_c \qquad (5.31)$$

i.e.

$30,9 = (a+s)/(a+s+1).36,0$, i.e., $(a+s) = 6,0$; for example $a = 5$ and $s = 1$. For these recycles the effluent nitrate concentration is calculated as

$$N_{ne} = N_c - D_{pl} = N_c/(a+s+1) = 5,1 \text{ mg } N.\ell^{-1}$$

This value for the effluent nitrate concentration is the lowest that can be obtained under the given circumstances, because $f_{x1} = f_M$ (the maximum allowable value) and the maximum denitrification potential thus created ($D_{pl} = (\alpha + K_2 C_r f_M)S_{bi}$) is fully utilized for nitrate removal. Consequently the minimum soluble effluent nitrogen concentration for optimal nitrogen removal is given by:

$$N_t = N_{ne} + N_a + N_o = 5,1 + 2,0 + 1,0 = 8,1 \text{ mg } N.\ell^{-1}$$

4.1 Influence of Temperature

The temperature of the mixed liquor has a very marked effect on nitrification and denitrification in single sludge systems. To

illustrate this, in Fig 5.10 the nitrification and denitrification capacities as well as the nitrates available in a pre-denitrification reactor of an MLE configuration for a = 3 and s = 1 are shown plotted as a function of the anoxic sludge mass fraction, f_x, for the same conditions as in Fig 5.8 except that the temperature is 20°C, i.e. $\mu_{nm20} = 0,36\ d^{-1}$ and $K_2 = 0,0042$ mg N.mg $X_a^{-1}h^{-1}$.

From the requirement for efficient nitrification the anoxic sludge fraction now can be $f_{x1} < f_M = 0,63$ (constraint 1 of Sec.2.3.2) but from the requirement for settleability $f_{x1} < f_{max} = 0,5$ (constraint 2 Sec. 2.3.2). For the anoxic sludge mass fraction $f_{x1} = 0,5$ the denitrification potential now not only exceeds the nitrates available for denitrification, N_{avl}, for a = 3 and s = 1, but is even considerably larger than the nitrification capacity. Hence in this case complete denitrification is feasible, but this cannot be achieved in a pre-denitrification system - it is necessary to introduce a post-denitrification reactor. The optimal division of the total anoxic sludge mass fraction between the pre- and post-denitrification reactors in order to obtain maximum (in this case complete) nitrate removal is discussed later in Section 5.

If the system has an anoxic sludge mass fraction of $f_{x1} = f_M$ (14°C) = 0,42 (thus being able to operate both at 14°C and at 20°C) the optimal recycle ratio can be found by again equating the denitrification capacity to the nitrate available.

Assuming the same values for N_s and N_o as at 14°C i.e. $N_s = 11,0$ and $N_o = 1,0$ mg $N.\ell^{-1}$, the nitrification potential is again $N_p = 50-11-1 = 38$ mg $N.\ell^{-1}$ and the nitrification capacity (Eq (5.15)) is $N_c = N_p - N_a = 38-0,8 = 37,2$ mg $N.\ell^{-1}$. The denitrification potential for $f_{x1} = 0,42$ is $D_{pl} = (0,028+0,1.\ 1,552.0,42)\ 400 = 37,3$ mg $N.\ell^{-1}$. Hence at 20°C the denitrification potential D_{pl} for $f_{x1} = 0,42$ is slightly higher than the nitrification capacity so that for any values of 'a' and 's', complete denitrification will take place in the anoxic reactor. The minimum effluent nitrate concentration with the Modified Ludzack Ettinger system under these circumstances is given by:

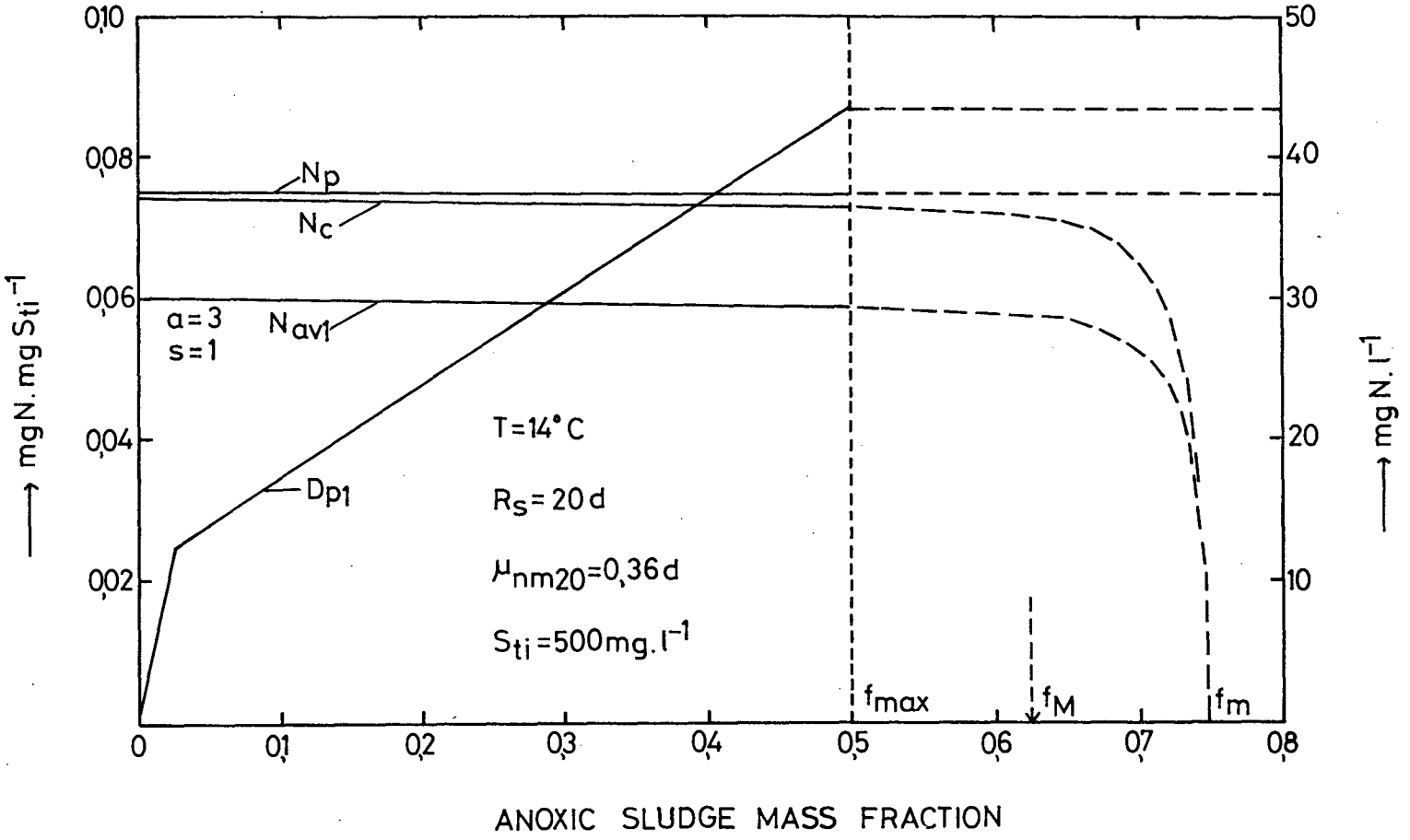

Fig. 5.10 *Nitrification potential (N_p) nitrification capacity(N_c), denitrification potential (D_p) and nitrate available for denitrification (N_{avl} for a=3 and s=1) as a function of the anoxic sludge mass fraction in a predenitrification system at 20°C.*

$$N_{ne} = N_c/(a+s+1)_{max}$$

where index max refers to the maximum values for the recycle ratios both 'a' and 's' recycle ratios.

4.2 Influence of sludge age

Both the nitrification capacity and the denitrification potential depend upon sludge age. The nitrification capacity increases with increasing sludge age because less influent TKN is incorporated in the sludge so that more can be nitrified, However, the increase of the nitrification capacity with increasing sludge age is relatively small. In contrast the denitrification potential, D_{pl}, increases rapidly with increasing sludge age because both the maximum anoxic sludge mass fraction, f_M, and the sludge mass factor, C_r, (Eq 5.18a) increase up to the sludge age R_{so} where f_M attains its maximum value: $f_M = f_{max} = 0,5$. An increase of the sludge age beyond the minimum for $f_M = f_{max}$, i.e. beyond R_{so}, results in only a moderate increase of the denitrification potential. To illustrate the points above in Fig 5.11 the nitrification capacity and denitrification potential of a pre-denitrification system are shown plotted as function of sludge age for $T = 14^oC$; $\mu_{nm14} = 0,18\ d^{-1}$; $S_{ti} = 500\ mg.\ell^{-1}$; $S_{bi} = 400\ mg.\ell^{-1}$; $N_{ti} = 50\ mg.\ell^{-1}$; $S_f = 1,25$.

From Fig 5.11 it is clear that from the point of nitrogen removal there is little merit in designing a system with a sludge age $R_s > R_{so}$.

The value of R_{so} itself is a function of μ_{nmT}: from (Eq 5.13b) substituting $f_{max} = f_M$

$$R_{so} = S_f/((1-f_{max})\mu_{nmT} - S_f.b_{nT}) \qquad (5.32)$$

In Fig 5.12 R_{so} is shown plotted as a function of the maximum specific growth rate, μ_{nm}, for temperatures $T = 14^oC$ and $T = 20^oC$ and for $f_M = f_{max} = 0,5$. For particular values of μ_{nm} and T the sludge age, R_{so}, can

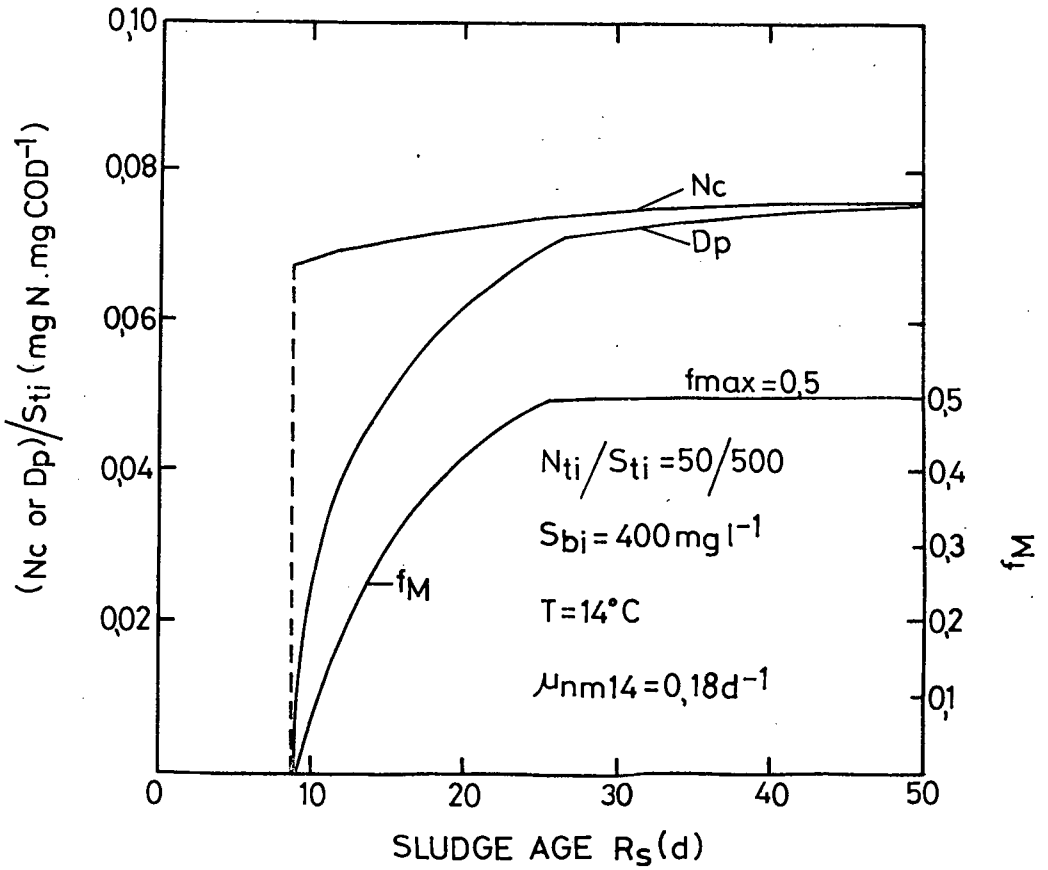

Fig. 5.11 *Nitrification capacity, N_c, denitrification potential of a predenitrification system, D_{pl}, and maximum anoxic sludge mass fraction, f_M, as a function of the sludge age ($f_{max} = 0,5$).*

be read off from Fig. 5.12 and the corresponding denitrification potential can be calculated from (Eq 5.27). If this denitrification potential exceeds the nitrification capacity then complete denitrification may be possible. If the denitrification potential for $R_s = R_{so}$ is less than the nitrification capacity, complete denitrification is <u>not</u> possible. Furthermore an increase of the sludge age beyond R_{so} will only marginally increase the denitrificaton potential. Hence for all practical purposes, the maximum extent of nitrate removal is set by the condition that $f_M = f_{max}$, (in this case $f_{max} = 0{,}5$) for the selected value of μ_{nmT}.

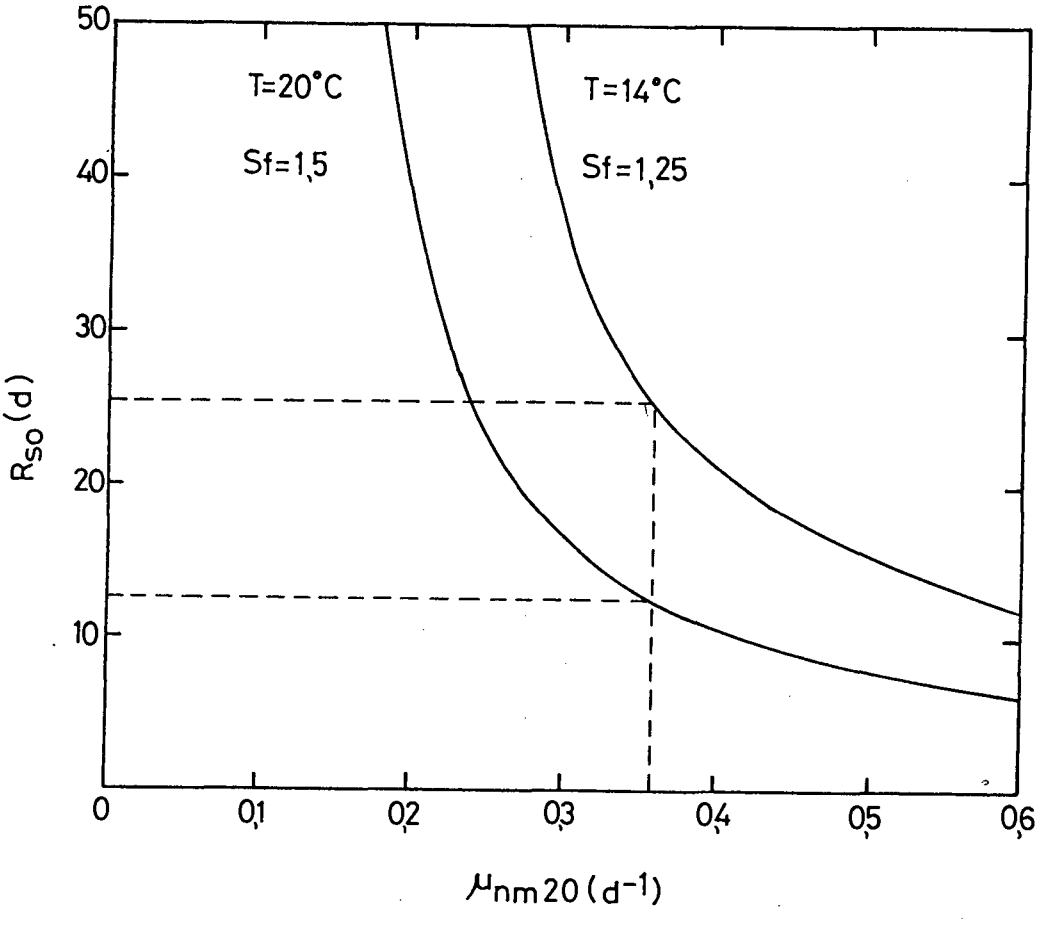

Fig. 5.12 *Minimum anoxic sludge age, R_{so}, required for the largest anoxic sludge mass fraction, f_{max}, as a function of the maximum specific growth rate of nitrifiers, μ_{nmT}, for a specified effluent ammonia concentration of 2 mg N.ℓ^{-1}. at T = 20°C (S_f = 1,5) and at T = 14°C(S_f = 1,25).*

5. OPTIMIZATION

The objective in optimization of nitrification-denitrification single sludge systems is to minimize the total soluble effluent nitrogen concentration, N_t. The concentration N_t is composed of ammoniacal, organic and nitrate nitrogen (the nitrite concentration usually is negligibly small). Usually also,both the ammonia and organic nitrogen concentrations in the effluent are very small in systems that have been designed for efficient nitrification whereupon these two nitrogen fractions remain practically constant in value in the range of an anoxic sludge mass fractions $0 < f_x < f_M$. Consequently, the optimization of nitrification-denitrification systems amounts to a minimization of the effluent nitrate concentration, N_{ne}, by maximizing the system nitrate removal.

From Fig 5.6 and Eqs(5.25 and 5.26) it is evident that the denitrification potential per unit anoxic reactor volume, i.e. the denitrification rate of a pre-denitrification reactor is greater than that of a comparable post-denitrification reactor. Hence , in any system always a pre-denitrification reactor is indicated for efficient use of the anoxic reactor volume. Thus the selection of the configuration for optimal nitrogen removal is limited to two options: a pre-denitrification reactor only, i.e. a Modified Ludzack-Ettinger process (Fig 5.1a) or pre- and post-denitrification reactors, i.e. a Bardenpho process (Fig 5.1c). Because of the higher rate of denitrification in a pre-denitrification reactor, for maximum nitrate removal it is necessary that as high a concentration of nitrate as possible is removed in this reactor. This is achieved by making the recycle ratios 'a' and 's' equal to the maximum allowable from a practical point of view and adapting the size of the pre-denitrification reactor to the recycled nitrates such that the nitrates available to the pre-denitrification reactor is equal to its denitrification potential, $N_{avl} = D_{pl}$. This sets the required sludge mass fraction in the pre-denitrification reactor and from the difference between the maximum anoxic sludge mass fraction, f_M, and the required sludge fraction in the pre-denitrification reactor, f_{x1}, the anoxic sludge mass fraction in the post-denitrification reactor, f_{x3}, is calculated. For waste flows with a high N_{ti}/S_{ti} ratio only a pre-denitrification reactor is indicated but now it is possible that the nitrate available to the pre-denitrification reactor, N_{av1}, at

the maximum recycle ratios 'a' and 's' exceeds the maximum denitrification potential than can be created in the pre-denitrification reactor (i.e. for $f_{x1} = f_M$). In this event the recycle ratios may be reduced until again $N_{av1} = D_{pl}$.

If a process is designed for efficient nitrification (i.e. $f_x \leq f_M$), nitrification capacity varies as the influent TKN concentration, N_{ti}, varies. In contrast the denitrification capacity varies as the influent COD concentration, S_{ti}, varies. Hence the N_{ti}/S_{ti} ratio is a rough measure of the ratio nitrification capacity/denitrification potential and the propensity of a process to achieve complete denitrification correspondingly is roughly indicated by the N_{ti}/S_{ti} ratio. Basically three different situations may arise:

1. Low N_{ti}/S_{ti} ratio. In this case complete denitrification is possible by using a pre- and post-denitrification system.

2. Medium N_{ti}/S_{ti} ratio. In this case denitrification is not complete but optimal removal requires both pre- and post-denitrification reactors.

3. High N_{ti}/S_{ti} ratio. In this case complete denitrification is not possible and optimal nitrate removal is attained by having a pre- denitrification reactor only.

5.1 Low N_{ti}/S_{ti} ratios

When complete denitrification is possible then for maximum efficiency in utilization of the anoxic sludge mass fraction, complete denitrification in both pre- and post-denitrification reactors must be achieved. The conditions under which this is possible are derived as follows: in the Bardenpho process, a fraction a/(a+s+1) of the nitrates produced in the aeration reactor (i.e. $N_c.a/(a+s+1)$) is recycled to the pre-denitrification reactor and the remainder (i.e. $N_c(s+1)/(a+s+1)$) is conveyed to the post-denitrification reactor. For complete denitrification in the pre-denitrification reactor:

$$D_{p1} = N_{av1}$$

or

$$(\alpha+K_2C_rf_{x1})S_{bi} = N_c a/(a+s+1) \quad (5.33)$$

i.e.,

$$f_{x1} = ((N_c/S_{bi})a/(a+s+1) - \alpha)/(K_2C_r) \quad (5.34)$$

and in the post-denitrification reactor:

$$D_{p3} = N_{av3}$$

$$K_3C_rf_{x3}S_{bi} = N_c(s+1)/(a+s+1) \quad (5.35)$$

i.e.

$$f_{x3} = ((N_c/S_{bi})(s+1)/(a+s+1))/(K_3C_r) \quad (5.36)$$

In both (Eqs 5.34 and 5.36) a and s are the maximum allowable recycle ratios. There are two constraints on the system: (1) The pre-denitrification reactor must be sufficiently large to allow complete utilization of easily biodegradable material (i.e. $f_{x1} \geq f_{min}$) and (2) The sum of the anoxic sludge fractions must not exceed the maximum allowable value for efficient nitrification f_M, i.e. $f_{x1} + f_{x3} \leq f_M$. Except in very unusual circumstances the first constraint is never operative, i.e. f_{x1} is always greater than f_{min}. The second constraint is important as it determines the <u>maximum</u> ratio of (nitrification capacity/ biodegradable influent concentration) for which complete denitrification is possible, $(N_c/S_{bi})_o$. This ratio can be determined by adding (Eqs 5.34 and 5.36) and equating the sum to f_M,

$$f_M = f_{x1} + f_{x3} = (a/(a+s+1))(N_c/S_{bi})_o - \alpha)/(K_2C_r) + ((s+1)/(a+s+1)) \cdot (N_c/S_{bi})_o/(K_3C_r) \quad (5.37)$$

i.e.

$$(N_c/S_{bi})_o = \frac{\alpha + K_2 C_r f_M}{a/(a+s+1) + (K_2/K_3)(s+1)/(a+s+1)} \qquad (5.38)$$

It is convenient to express (Eq 5.38) in terms of the N_{ti}/S_{ti} ratio as this ratio is directly available as the influent TKN/COD ratio: For the maximum $(N_c/S_{bi})_o$ ratio the corresponding $(N_{ti}/S_{ti})_o$ ratio can be calculated as follows:

$$N_c/S_{bi} = (N_{ti}-N_s-N_o-N_a)/((1-f_{us}-Pf_{up})S_{ti})$$

$$= N_{ti}/(S_{ti}(1-f_{us}-Pf_{up})) - (N_s+N_o+N_a)/(S_{ti}(1-f_{us}-Pf_{up})) \qquad (5.39)$$

In a particular design situation the value $(N_s+N_o+N_a)/(S_{ti}(1-f_{us}-Pf_{up}))$ is known. For raw sewage this value usually is about 0,03 mg TKN. mg COD^{-1} so that approximately:

$$N_{ti}/S_{ti} = (1-f_{us}-Pf_{up})N_c/S_{bi} + 0,03 \qquad (5.40)$$

Hence

$$(N_{ti}/S_{ti})_o = \frac{(1-f_{us}-Pf_{up})(\alpha+K_2 C_r f_M)}{\frac{a}{a+s+1} + \frac{K_2}{K_3}\cdot\frac{s+1}{a+s+1}} + 0,03 \qquad (5.41)$$

If the N_{ti}/S_{ti} ratio of a waste flow is less than the value calculated from Eq(5.41) then complete denitrification is feasible and it is possible either to reduce the sludge age until Eq(5.41) is satisfied or to reduce the anoxic sludge mass fractions $(f_{x1} + f_{x3})$. In the latter event the minimum anoxic sludge mass fractions f_{x1} and f_{x3} that allow complete denitrification are calculated directly from Eqs(5.34 and 5.36) respectively; both f_{x1} and f_{x3} decrease linearly with decreasing (N_c/S_{bi}) ratios for $(N_c/S_{bi}) < (N_c/S_{bi})_o$.

5.2 Medium N_{ti}/S_{ti} ratio

In terms of (Eq 5.41) with maximum recycle ratios (a+s), if complete denitrification is not possible the optimal division of the

anoxic sludge mass fraction between pre- and post-denitrification reactors can be determined as follows: in the pre-denitrification reactor the nitrate removal per unit volume is always greater than in the post-denitrification reactor. Hence for maximal nitrate removal the pre-denitrification reactor must be made sufficiently large to remove all the nitrates recycled to it at maximum recycle (a+s). With an effluent nitrate concentration, N_{ne}, (equal to the nitrate concentration in the post-denitrification reactor) the nitrate available to the pre-denitrification reactor is:

$$N_{avl} = sN_{ne} + a\,N_2 \tag{5.42}$$

Where N_2 = nitrate concentration in the aerobic reactor.

If f_{x1} can be made such that $D_{p1} = N_{avl}$ then the nitrate concentration in the pre-denitrification reactor, N_1, will be equal to zero and the nitrate concentration in the aerobic reactor $N_2 = N_c/(a+s+1)$. Hence the nitrate concentration in the post-denitrification reactor is given by:

$$N_{ne} = N_2 - D_{p3}/(s+1) = N_c/(a+s+1) - D_{p3}/(s+1) \tag{5.43}$$

Substituting (Eq 5.43) in (Eq 5.42)

$$N_{avl} = D_{p1} = (a+s)/(a+s+1).N_c - s/(s+1).D_{p3} \tag{5.44}$$

Substituting for D_{p1} and D_{p3} from (Eqs 5.18 and 5.19) respectively and for f_{x3} from (Eq 5.37)

$$f_{x1} = \frac{\frac{a+s}{a+s+1}\frac{N_c}{S_{bi}} - \alpha - \frac{s}{s+1}K_3 C_r f_M}{C_r\left(K_2 - \frac{s}{s+1}\,K_3\right)} \tag{5.45}$$

and

$$f_{x3} = f_M - f_{x1} \tag{5.46}$$

If f_{x1} calculated from (Eq 5.45) is less than f_M, Eqs (5.45 and 5.46)

apply and define the anoxic sludge mass fractions in the pre- and post-denitrification reactors respectively. If f_{x1} is greater than f_M then f_{x3} theoretically is a negative quantity. This implies that it is not possible to remove all the nitrates in the pre-denitrification reactor, i.e. the post-denitrification reactor must be omitted, and the entire anoxic sludge mass fraction located in the pre-denitrification reactor: $f_{x1} = f_M$. The ratio (N_c/S_{bi}) at which f_{x1} becomes equal to f_M, $(N_c/S_{bi})_1$, i.e. when a post-denitrification reactor becomes counterproductive, can be calculated from Eq(5.45) by equating f_{x1} to f_M, i.e.

$$(N_c/S_{bi})_1 = (\alpha+K_2 C_r f_M)(a+s+1)/(a+s) \qquad (5.47)$$

The ratio $(N_c/S_{bi})_1$ defines the upper limit for having a post-denitrification reactor in the system. The ratio $(N_{ti}/S_{ti})_1$ associated with $(N_c/S_{bi})_1$ can be found from Eq(5.39)

5.3 High N_{ti}/S_{ti} ratio

If the actual (N_c/S_{bi}) ratio of a waste flow is greater than the value $(N_c/S_{bi})_1$ (calculated from Eq(5.47)) then (a) only a pre-denitrification reactor is present in the system with $f_{x1} = f_M$, and (2) the possibility exists that the maximum (a+s) is reduced to a value such that the nitrate in the effluent from the anoxic reactor just becomes zero, i.e. the nitrate available is made equal to the denitrification potential. The optimal recycle ratios can be found as follows: The 's'-recycle normally will be kept at the minimum value required for proper operation of the settling tank so that only the a-recycle is adjustable. The appropriate 'a'-recycle is found from $N_{av1} = D_{p1}$, i.e.

$$N_c(a+s)/(a+s+1) = (\alpha + K_2 C_r f_M) S_{bi} \qquad (5.50)$$

$$a = \frac{(s+1)(\alpha+K_2 C_r f_M) - s\, N_c/S_{bi}}{(N_c/S_{bi}) - (\alpha+K_2 C_r f_M)} \qquad (5.51)$$

At very high (N_c/S_{bi}) ratios it is possible that 'a' in (Eq 5.51) becomes negative. If so the 'a'-recycle becomes redundant and only the 's'-recycle is needed. The (N_c/S_{bi}) ratio at which the 'a'-recycle becomes zero, $(N_c/S_{bi})_2$, is calculated from (Eq 5.51):

$$(N_c/S_{bi})_2 = (\alpha + K_2 C_r f_M)(s+1)/s \qquad (5.52)$$

At (N_{ti}/S_{ti}) ratios greater than that associated with $(N_c/S_{bi})_2$ (through Eq(5.39)) the 's'-recycle alone introduces sufficient or more than sufficient nitrate into the predenitficiation reactor to satisfy the denitrification potential.

From the analysis above it is clear that the (N_{ti}/S_{ti}) ratio in the influent has a profound effect on the design of a single sludge nitrification-denitrification system. To illustrate the manner in which the (N_c/S_{bi}) or the (N_{ti}/S_{ti}) ratio affects the design it is convenient to accept constant values for flow, influent COD concentration, sewage characteristics (f_{us}, f_{up}, α , μ_{nmT}, T) and operational conditions (R_s, a, s) and analyse the influence of a changing (N_{ti}/S_{ti}) or (N_c/S_{bi}) ratios on the optimal reactor configuration and the minimal effluent nitrate concentration. Accept the same assumptions as in the previous examples, i.e. T = 14°C; R_s = 20 d; $\mu_{nmT} = 0{,}18\ d^{-1}$; $f_{ca} = 0{,}24$; $N_o = 1$ mg N.ℓ^{-1}; $N_s = 11$ mg N.ℓ^{-1} and maximum recycle ratios of a = 4 and s = 1. The limiting ratios $(N_c/S_{bi})_o$; $(N_c/S_{bi})_1$ and $(N_c/S_{bi})_2$ are calculated by means of Eqs(5.38, 5.47 and 5.52) respectively. For a maximum anoxic sludge mass fraction $f_M = 0{,}40$:

$$(N_c/S_{bi})_o = 0{,}075 \text{ mg N.mg COD}^{-1}$$

$$(N_c/S_{bi})_1 = 0{,}090 \text{ mg N.mg COD}^{-1}$$

$$(N_c/S_{bi})_2 = 0{,}150 \text{ mg N.mg COD}^{-1}$$

The corresponding (N_{ti}/S_{ti}) ratios can be calculated from Eq(5.39):

With $(1-f_{us}-P.f_{up}) = 0,8$ and

$(N_s+N_o+N_a)/(S_{ti}(1-f_{us}-P.f_{up})) = 0,03$ mg N.mg COD^{-1}.

$(N_{ti}/S_{ti})_o = 0,09$

$(N_{ti}/S_{bi})_1 = 0,10$

$(N_{ti}/S_{ti})_2 = 0,15$

In the example the anoxic sludge mass fractions f_{x1}, and f_{x3} for complete denitrification of the maximum ratio $(N_c/S_{bi})_o$ are calculated from Eqs(5.34 and 5.36) respectively: $f_{x1} = 0,19$ and $f_{x3} = 0,21$. For all (N_c/S_{bi}) ratios less than $(N_c/S_{bi})_o$ complete denitrification can be achieved with anoxic sludge mass fractions less than $f_{x1} = 0,19$ and $f_{x3} = 0,21$. In this range of (N_c/S_{bi}) ratios, minimum anoxic sludge fractions required for complete nitrate removal can be calculated using Eqs(5.34 and 5.36) for the pre- and post-denitrification reactor respectively. From Eqs(5.34 and 5.36) there is a linear relationship between the required anoxic sludge mass fractions f_{x1} or f_{x3} and the (N_c/S_{bi}) ratio. However, there is a minimum anoxic sludge mass fraction in the pre-denitrification reactor, set by the requirement that the utilization of easily biodegradable material must be complete in this reactor, i.e. $f_{x1} \geq f_{min}$. The value of f_{min} has been established earlier from Eq (2.76): $f_{min} = \alpha/(K_1 C_r)$.

The anoxic sludge mass fractions f_{x1} and f_{x3} in the range $(N_c/S_{bi})_o \leq (N_c/S_{bi}) \leq (N_c/S_{bi})_1$ are calculated from Eqs(5.45 and 5.46) respectively. From Eq (5.45), in this range there is a linear increase of f_{x1} with an increasing (N_c/S_{bi}) ratio and correspondingly f_{x3} decreases linearly with increasing (N_c/S_{bi}) ratios (Eq(5.46)) until $f_{x3} = 0$ when $(N_c/S_{bi}) = (N_c/S_{bi})_1$. For any (N_c/S_{bi}) ratio greater than $(N_c/S_{bi})_1$, always $f_{x1} = f_M$ and $f_{x3} = 0$.

In Fig 5.13a the values of f_{x1} and f_{x3} are shown plotted as a function of (N_c/S_{bi}) or (N_{ti}/S_{ti}). The plot graphically depicts the different ranges of (N_c/S_{bi}) ratios (and hence of (N_{ti}/S_{ti}) ratios) in which the different configurations are operative. Once the system

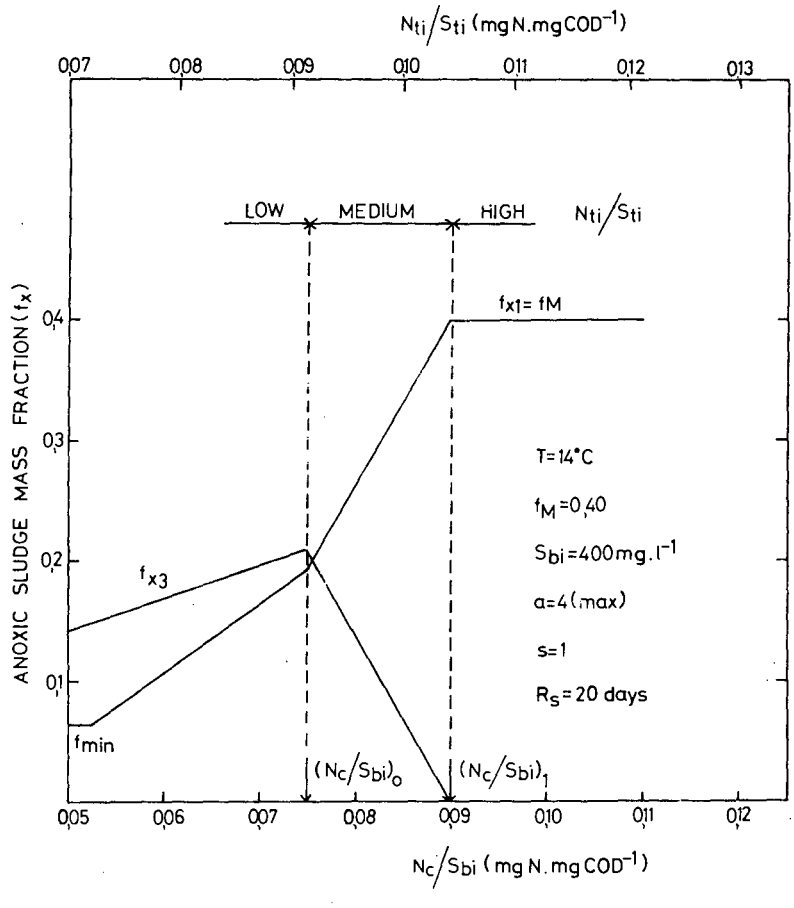

Fig.5.13a *Optimal anoxic sludge mass fractions f_{x1} and f_{x3} as a function of the ratio(N_c/S_{bi}) (bottom scale) and N_{ti}/S_{ti} (top scale).*

configuration has been determined, the nitrate concentration in the effluent is readily calculated: From the basis of selection of the anoxic sludge mass fractions of the pre- and post-denitrification reactors, at any (N_c/S_{bi}) ratio less than $(N_c/S_{bi})_o$, the effluent nitrate concentration is zero, because the reactors are designed to give complete denitrification. For all (N_c/S_{bi}) ratios greater than $(N_c/S_{bi})_o$ the effluent concentration is calculated by considering that in this range the nitrate removal in the anoxic reactors is maximum, i.e. equal to the denitrification potentials, Hence:

$$N_{ne} = N_c - D_{p1} - D_{p3} \qquad (5.53)$$

The nitrate concentration, N_{ne}, is shown plotted as a function of the ratio (N_c/S_{bi}) in Fig 5.13b for the same conditions as for the plot in Fig 5.13a.

Insofar as the optimal recycle ratio 'a' and 's' are concerned, (a+s) must be the maximum selected for all $(N_c/S_{bi}) < (N_c/S_{bi})_1$ and a = 0 for $(N_c/S_{bi}) > (N_c/S_{bi})_2$. In the range $(N_c/S_{bi})_1 < (N_c/S_{bi}) < (N_c/S_{bi})_2$, the 'a'-recycle is determined by Eq(5.51); in Fig 5.13c the optimal 'a'-recycle is plotted as a function of the (N_c/S_{bi}) ratio for the same conditions as for the plot in Fig 5.13a.

5.4 Relationship between sludge age, reactor configuration and extent of nitrogen removal

In the above Section a constant sludge age of R_s = 20 d was assumed. In this Section this restriction will be relinquished and the influence of sludge age on the reactor configuration and extent of nitrogen removal is analysed, still keeping the sewage characteristics and operational conditions the same as in the previous sections, i.e.

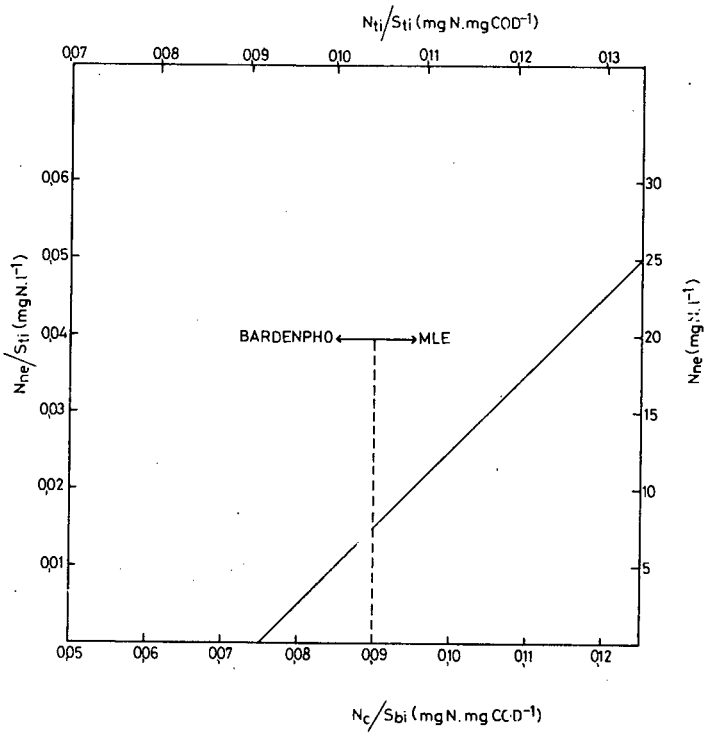

Fig. 5.13b *Minimum effluent nitrate concentration, N_{ne}, as a function of the ratio (N_c/S_{bi}) (bottom scale) and (N_{ti}/S_{ti}) (top scale). For sewage characteristics and operational conditions, see Fig. 5.13a.*

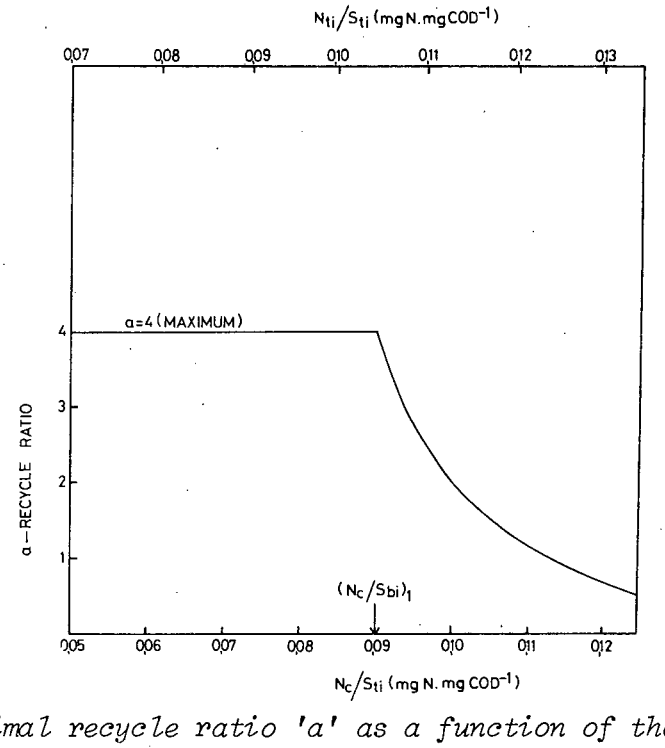

Fig. 5.13c *Optimal recycle ratio 'a' as a function of the (N_c/S_{bi}) ratio (bottom scale) and the (N_{ti}/S_{ti}) ratio (top scale). For sewage characteristics and operational conditions see Fig. 5.13a.*

S_{ti} = 500 mg.ℓ^{-1}; S_{bi} = 400 mg.ℓ^{-1}; f_{ca} = 0,24; T = 14°C; μ_{nm14} = 0,18 d^{-1}; s_f = 1,25; a = 4 and s = 1. Equation (5.37) can be used to determine the minimum sludge age for complete denitrification at any particular (N_c/S_{bi}) ratio. Rearranging Eq (5.37):

$$C_r f_M = ((a/a+s+1) + (K_2/K_3)(s+1)/(a+s+1)(N_c/S_{bi})_o - \alpha)/K_2 \qquad (5.54)$$

Substituting for $C_r = Y_h R_s/(1+b_{hT}R_s)$ and for $f_M = 1-S_f(b_{nT}+1/R_s)/\mu_{nmT}$:

$$C_r f_M = Y_h R_s\ (\mu_{nmT}-S_f(1+b_{nT}))/(\mu_{nmT}(1+b_{hT}R_s)) \qquad (5.55)$$

Equation (5.55) is only valid if $f_M < f_{max}$. For $f_M = f_{max}$ (i.e. for $R_s \geq R_{so}$):

$$C_r f_M = Y_h R_s f_{max}/(1+b_{hT}R_s)\ \ (f_M = f_{max}) \qquad (5.55a)$$

Substituting Eqs (5.55 or 5.55a) in Eq (5.54) gives an implicit expression for the minimum sludge age required for complete denitrification as a function of (N_c/S_{bi}), or alternatively, the maximum (N_c/S_{bi}) ratio that can be denitrified completely in a Bardenpho process at a particular sludge age, i.e. $(N_c/S_{bi})_o$. In Fig 5.14 the value of $(N_c/S_{bi})_o$ is plotted as a function of sludge age.

The plot is valid only in the range where the sludge age is sufficient to allow complete utilization of the easily biodegradable influent material in the pre-denitrification reactor; if the sludge age is reduced sufficiently, a point is reached where f_{x1} is too small for complete utilization of the easily biodegradable material. The minimum sludge age where this problem arises is when the anoxic sludge mass fraction in the pre-denitrification reactor f_{x1} becomes equal to $f_{min} = \alpha/(K_1 C_r)$ (Eq(2.76)). The associated value of the anoxic sludge mass fraction in the post-denitrification reactor, f_{x3}, is determined as follows: at $f_{x1} = f_{min}$, for complete removal of nitrate, a fraction a/(a+s+1) of the nitrification capacity must be removed in the pre-denitrification reactor and a fraction (s+1)/(a+s+1) in the post-denitrification reactor, i.e.

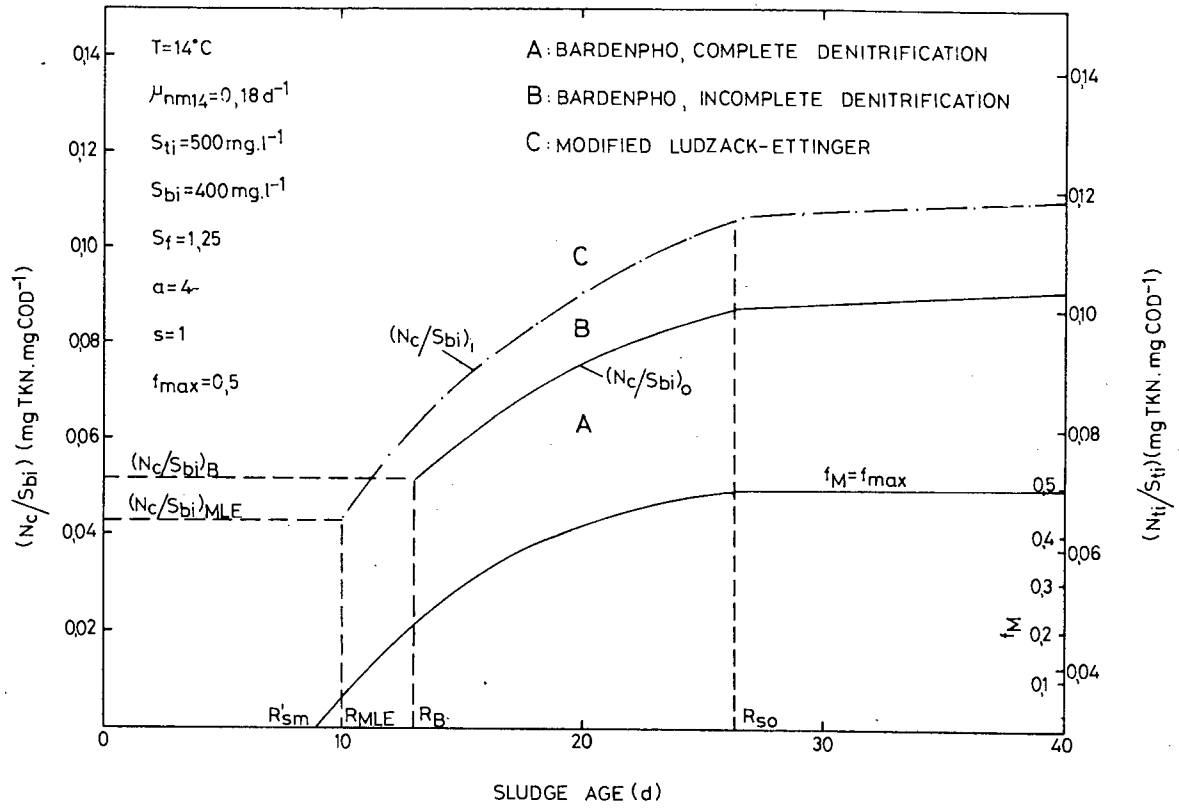

Fig. 5.14 *N_c/S_{bi} ratio for complete denitrification, $(N_c/S_{bi})_o$, and for transition of a Bardenpho to a Modified Ludzack Ettinger configuration, $(N_c/S_{bi})_1$, as a function of sludge age (corresponding N_{ti}/S_{ti} ratios on right hand scale). The maximum anoxic sludge mass fraction f_M is also plotted as a function of sludge age.*

$$D_{p1} = (K_1+K_2)C_r f_{min} S_{bi} = N_c.a/(a+s+1) \quad (5.56)$$

$$D_{p3} = K_3 C_r f_{x3} S_{bi} = N_c.(s+1)/(a+s+1) \quad (5.57)$$

The required sludge mass fraction in the post-denitrification reactor is found by dividing (Eq 5.56) by (Eq 5.57):

$$f_{x3} = ((K_1+K_2)/K_3)\ ((s+1)/a).f_{min} \quad (5.58)$$

Hence the minimum total anoxic sludge mass fraction, $f_{min,B}$ in a Bardenpho configuration with complete nitrate removal is given by:

$$f_{min,B} = f_{x1} + f_{x3}$$

$$= f_{min} + ((K_1+K_2)/K_3)((s+1)/a).f_{min}$$

$$= (1+((K_1+K_2)/K_3)((s+1)/a))\ .f_{min}$$

The sludge age associated with the anoxic sludge mass fraction $f_{min,B}$, R_B, is now found by equating $f_{min,B}$ to the maximum anoxic sludge mass fraction f_M (Eq 5.13b) $f_{min,B} = f_M$.

or

$$(1+((K_1+K_2)/K_3)((s+1)/a))f_{min} = 1-S_f(b_{nT}+1/R_B)/\mu_{nmT} \quad (5.60)$$

where

R_B = minimum sludge age of a Bardenpho configuration with complete nitrate removal.

$f_{min} = \alpha/K_1 C_r) = \alpha(1+b_{hT} R_B)/(K_1 Y_h R_B)$

Eq(5.60) cannot be solved for R_B, but a solution can be obtained by trial and error. For the given conditions, R_B = 13 d and this is the lowest value for the sludge age that allows utilization of the Bardenpho configuration for the conditions in Fig 5.14. Numerically the values of the anoxic sludge mass fractions at $R_s = R_B$ are: $f_{x1} = f_{min}$ = 0,07

and $f_{x3} = ((K_1+K_2)/K_3)((s+1)/a).f_{min} = 0,17$. The denitrification potential for the system is $D_p = D_{p1} + D_{p3}$. Using Eqs (5.56 and 5.57): $D_p = 0,052\ S_{bi}$. The maximum (N_c/S_{bi}) ratio that allows complete denitrification at $R_s = R_B$ requires that $N_c = D_p$. Hence, under the given conditions, the (N_c/S_{bi}) ratio that allows complete denitrification at minimum sludge age R_B is given by:

$$(N_c/S_{bi})_B = 0,052 \text{ mg TKN.mg COD}^{-1} \qquad (5.61)$$

The minimum sludge age R_B and the corresponding maximum (N_c/S_{bi}) ratio for complete denitrification, $(N_c/S_{bi})_B$, are indicated in Fig 5.14. From Fig 5.14, in the range R_B (where $f_x = f_M = f_{min,B}$) to R_{so} (where $f_x = f_M = f_{max}$), the value of $(N_c/S_{bi})_o$ increases considerably with increasing sludge age. An increase of the sludge age beyond the value of R_{so} (which, for the given constants, is 26,3 days) results in an insignificant increase of the ratio $(N_c/S_{bi})_o$. This, from the point of view of nitrogen removal, once again shows that there is little merit in increasing the sludge age beyond the value R_{so}.

In a similar analysis as above the value of $(N_c/S_{bi})_1$, i.e. the (N_c/S_{bi}) ratio for which the post-denitrification $f_{x3} = 0$, or, the (N_c/S_{bi}) ratio at the transition point from a Bardenpho to an MLE configuration, can also be determined as a function of sludge age. By rearranging Eq(5.47),

$$C_r f_M = ((N_c/S_{bi})_1 (a+s)/(a+s+1) - \alpha)/K_2 \qquad (5.62)$$

By substituting for $C_r f_M$ from Eq(5.55) the value of $(N_c/S_{bi})_1$ can be calculated as a function of R_s. The relationship between $(N_c/S_{bi})_1$ and R_s is also shown plotted in Fig 5.14 for the same conditions as above, for $(N_c/S_{bi})_o$ versus R_s. The minimum sludge age for the pre-denitrification systen, R_{MLE}, is characterised by $f_M = f_{min}$, i.e.

$$1 - S_f(b_{nT}+1/R_{MLE})/\mu_{nmT} = \alpha(1+b_{hT}R_{MLE})/(K_1 Y_h R_{MLE}) \qquad (5.63)$$

For the given constants, by trial and error: R_{MLE} = 10 d. The corresponding denitrification potential $D_{pmin} = (\alpha + K_2 C_r f_{min}) S_{bi} = 0{,}036\ S_{bi}$. In the MLE configuration a fraction (a+s)/(a+s+1) of the nitrification capacity is recycled to the anoxic zone. Hence the nitrification capacity that allows complete denitrification in the anoxic reactor at $R_s = R_{MLE}$ is characterised by:

$$N_c = D_{pmin}\ (a+s+1)/(a+s)$$

or

$$(N_c/S_{bi})_{MLE} = \frac{a+s+1}{a+s}\ (\alpha + K_2 C_r f_{min}) = 0{,}043 \text{ mg N.mg COD}^{-1} \qquad (5.64)$$

where

$(N_c/S_{bi})_{MLE}$ = maximum (N_c/S_{bi}) ratio that gives complete denitrification in the anoxic reactor for $R_s = R_{MLE}$

The values of R_{MLE} and the corresponding $(N_c/S_{bi})_{MLE}$ are indicated in Fig 5.14.

From Fig 5.14 the shape of the curve for $(N_c/S_{bi})_1$ as a function of R_s is very similar to that for $(N_c/S_{bi})_o$ versus R_s. From the minimum sludge age, R_{MLE}, to R_{so}, for which $f_M = f_{max}$, there is a sharp increase of $(N_c/S_{bi})_1$ with increasing R_s, whereas an increase of R_s beyond R_{so} does not affect appreciably the corresponding $(N_c/S_{bi})_1$ value.

Figure 5.14 can now be divided into three global zones A, B and C (see Fig 5.14). In zone A the (N_c/S_{bi}) ratio is lower than $(N_c/S_{bi})_o$ i.e., complete removal of nitrate in a Bardenpho process is possible. In zone B, characterised by $(N_c/S_{bi})_o < (N_c/S_{bi}) < (N_c/S_{bi})_1$ there is incomplete denitrification in the post-denitrification reactor but complete denitrification in the pre-denitrification reactor. In both zone A and B the recycle ratios 'a' and 's' are maximum (in Fig 5.14: a = 4 and s = 1). In zone C the (N_c/S_{bi}) ratio is higher than $(N_c/S_{bi})_1$ i.e., there is no post-denitrification reactor and the Modified Ludzack Ettinger is the optimal process configuration; the denitrification potential in the pre-denitrification reactor is insufficient to remove the nitrate at the maximum recycle ratios 'a' and 's' and these may

be reduced until the nitrate available for denitrification becomes equal to the denitrification potential. (Keeping the 'a' and 's' recycles at the former maximum values does not reduce the nitrate removal efficiency - only now there will be nitrate in the effluent from the pre-denitrification reactor).

6. CYCLIC FLOW AND LOAD CONDITIONS

When the influent flow and/or the COD or TKN concentrations vary daily cyclicly the concepts of nitrification capacity and denitrification potential, as defined previously, become inadequate to describe the nitrification and denitrification processes, because the reaction rates of nitrification and denitrification also vary cyclicly. In order to assess nitrification and denitrification performance the nitrification capacity and denitrification potential under cyclic conditions may be redefined as:

$$N_c = \frac{\text{mass of nitrified TKN/day}}{\text{influent flow/day}} \quad (\text{mg N}.\ell^{-1}) \qquad (5.65)$$

and

$$D_p = \frac{\text{maximum mass of denitrified nitrate/day}}{\text{influent flow/day}} \quad (\text{mg N}.\ell^{-1}) \qquad (5.66)$$

The values of the nitrification capacity, N_c, and the denitrification potential, D_p, thus defined can be calculated only by using computer simulations. To analyse the influence of a cyclic flow and load patterns on nitrification and denitrification the response of sinusoidal input waves (flow, TKN and COD concentrations) with a frequency of one day was simulated. The flow, TKN and COD waves were chosen to be in phase (i.e. maximum flow and concentrations occur simultaneously) as this pattern is approximated in practice. Simulations were carried out for several fractional amplitudes of the influent flow and the influent TKN and COD concentrations, the fractional amplitude A being defined as (See Chapter 3, Section 10.2):

A = (maximum value - minimum value)/(2 x average value)

To assist in making comparisons for different fractional amplitudes the average values of flow, COD and TKN were chosen such that the flow and load per day were the same for all analysed fractional amplitudes.

6.1 Nitrification behaviour under cyclic flow and load conditions

Simulations were done using the general model to analyse the nitrification response of anoxic/aerobic systems to sinoidal input waves with different fractional amplitudes of flow (A_f), influent TKN concentrations (A_{TKN}) and influent COD concentrations (A_{COD}), and under a variety of operational conditions and sewage characteristics. Typical results of these simulations are shown in Fig 5.15, where the average effluent ammonia concentration from a pre-denitrification system is shown plotted, as a function of the anoxic sludge mass fraction for the following conditions: $T = 14^{o}C$; $R_s = 20$ d; mean $R_h = 0,5$ d; mean $S_{ti} = 500$ mg.ℓ^{-1}; mean $N_{ti} = 50$ mg.ℓ^{-1}; $\mu_{nm14} = 0,15$ d^{-1}. Plots are shown for $A_f = A_{TKN} = 0,0$ (i.e., constant flow and load conditions) $A_f = 0,5$ and $A_{TKN} = 0,0$; $A_f = 0,0$ and $A_{TKN} = 0,5$ and $A_f = A_{TKN} = 1,0$. (The value of A_{COD} has virtually no effect on the nitrification response).

From the simulations an increasing fractional amplitude of either flow (A_f) or of the TKN concentration (A_{TKN}) tended to cause an increase in the mean effluent ammonia concentration i.e., tended to reduce the nitrification capacity. From numerous simulations the ammonia effluent concentration for different fractional amplitudes of flow and load and for different average hydraulic retention times and sludge ages, an approximated empirical relationship between the mean ammonia effluent concentration under daily sinusoidally cyclic flow and load conditions, $N_{a,cy}$, and the ammonia effluent concentration under constant flow and load conditions, $N_{a,co}$, was found:

$$N_{a,cy} = [1+(A_f+A_{TKN})/R_h] \; N_{a,co} \qquad (5.67)$$

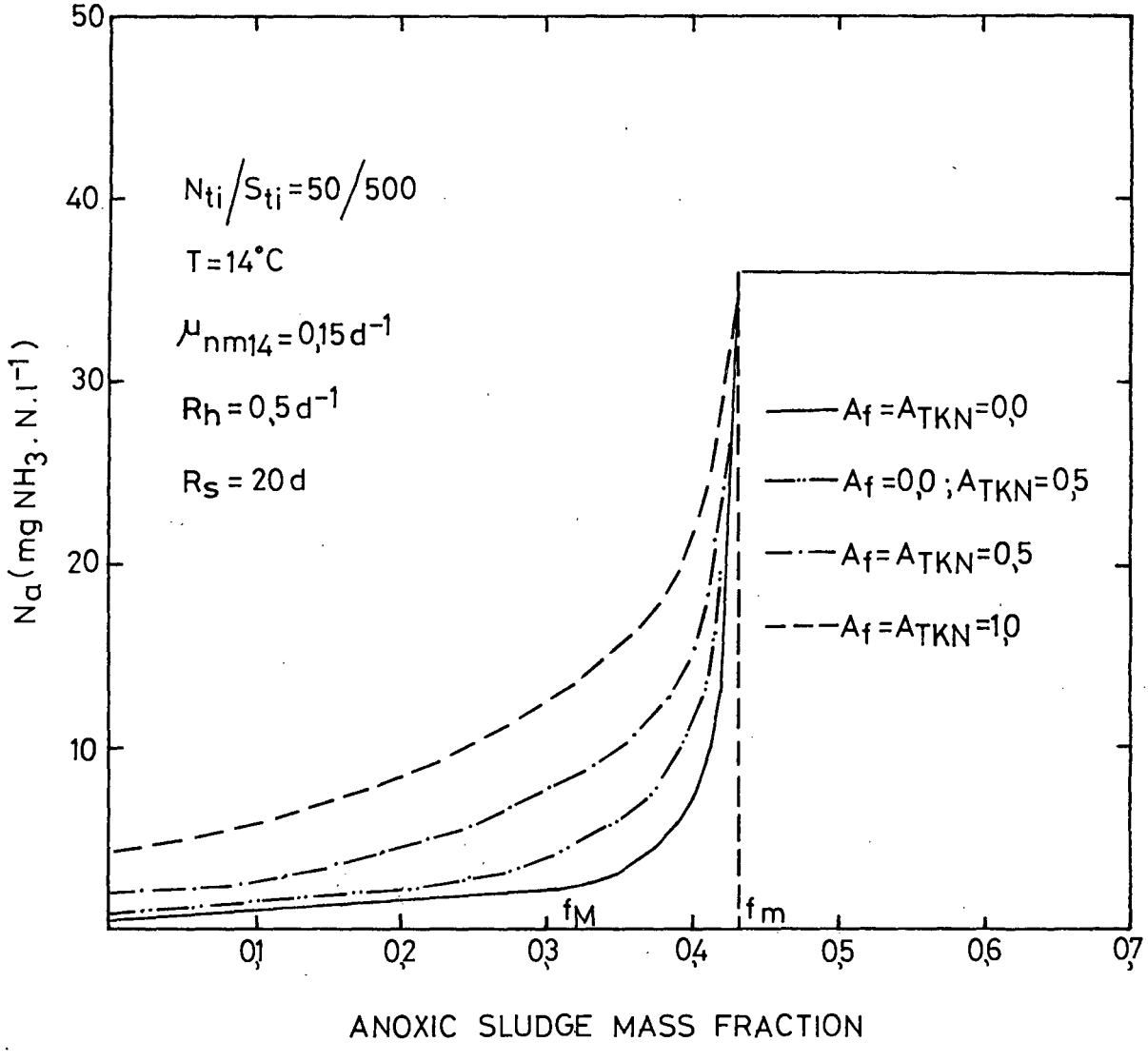

Fig. 5.15 *Mean ammonia effluent concentration from a predenitrification system for sinoidal input waves and influent TKN and COD concentrations as a function of the anoxic sludge mass fraction.*

where

$N_{a,cy}$ = approximated value of the mean effluent ammonia concentration under sinusoidal flow and load conditions (mg $N.\ell^{-1}$).

$N_{a,co}$ = effluent ammonia concentration under constant flow and load conditions (mg $N.\ell^{-1}$).

A_f = fractional amplitude of the sinusoidal flow.

A_{TKN} = fractional amplitude of the sinusoidal influent TKN concentration.

R_h = average hydraulic retention time (d).

The relationship of Eq(5.67) forms a good approximation for mean effluent ammonia concentrations, $N_{a,cy}$, less than about 15 mg $N.\ell^{-1}$ (see Fig 5.15) irrespective of other factors that influence nitrification behaviour such as sludge age, temperature, maximum specific growth rate of nitrifiers or anoxic sludge mass fraction. Eq(5.67) forms the basis for a design estimation of nitrification behaviour in anoxic/aerobic single sludge systems under cyclic flow and load conditions: For a specified mean effluent ammonia concentration, $N_{a,cy} = N_{ad}$, under cyclic flow and load conditions the minimum aerobic and maximum anoxic sludge mass fractions can be estimated as follows:

1. Establish the effluent ammonia concentration that would be obtained if the system was fed under constant flow and load conditions. From Eq(5.67):

$$N_{a,co} = N_{ad}/(1 + (A_f + A_{TKN})/R_h)$$

N_{ad} = specified desired average effluent ammonia concentration under cyclic flow and load conditions (mg $N.\ell^{-1}$).

$N_{a,co}$ = effluent ammonia concentration under constant flow and load conditions (mg $N.\ell^{-1}$).

2. Calculate the maximum anoxic sludge mass fraction giving an effluent ammonia concentration, $N_{a,co}$, under constant flow and load conditions.

(Eq 5.13a):

$$f_M = 1 - (1+K_{nT}/N_{a,co})(b_{nT}+1/R_s)/\mu_{nmT}$$

$$f_M = 1 - (1+K_{nT}((1+A_f+A_{TKN})/R_h)/N_{ad})(b_{nT}+1/R_s)/\mu_{nmT} \qquad (5.68)$$

A system with an anoxic sludge mass fraction, f_M, given by Eq(5.68) under cyclic flow and load conditions with fractional amplitudes A_f and A_{TKN} for influent flow and TKN concentration respectively will produce a mean effluent ammonia concentration, $N_{a,cy}$, close to the specified desired value N_{ad}. From Eq(5.68), the result of sinusoidally cyclic conditions of flow and/or TKN concentration is equivalent to an increase of the half saturation value K_{nT} by a factor $(1+A_f+A_{TKN})/R_h$, i.e. there is an increase in the required safety factor (Eq (5.12)):

$$S_{f,cy} = (1+(A_f+A_{TKN})/R_h) \cdot S_f = (1+(A_f+A_{TKN})/R_h)(1+K_{nT}/N_{ad}) \qquad (5.69)$$

$S_{f,cy}$ = required safety factor under cyclic flow and load conditions to produce an average effluent ammonia concentration N_{ad}.

S_f = required safety factor under constant flow and load conditions to produce N_{ad}.

The increase of the required safety factor with increasing values of A_f and A_{TKN} implies that for the same desired effluent ammonia concentration, N_{ad}, the maximum anoxic sludge mass fraction under cyclic flow and load conditions is always smaller than that under constant flow and load conditions. This in turn tends to reduce the extent of denitrification achievable under cyclic flow and load conditions. The reason for the increase in the effluent ammonia concentration (i.e., reduction of the nitrification efficiency) with increasing severity of cyclic flow and load conditions is that nitrification is a relatively slow reaction; the nitrifiers even operating at maximum rate only have a limited capacity to deal with a sudden increase of the influent TKN load.

6.2 Denitrification behaviour under cyclic flow and load conditions

Simulations were carried out, using the general model to analyse the denitrification behaviour of pre-denitrification systems under sinusoidal input waves with different fractional amplitudes of flow, A_f, influent TKN concentration, A_{TKN}, and influent COD concentration, A_{COD}. Fig 5.16 shows a plot of the denitrification potential in a pre-denitrification system (as defined in Eq (5.66)) as a function of the anoxic sludge mass fraction for sinusoidal input waves of flow and influent TKN and COD concentrations (all in phase). Fractional amplitudes of 0,0 (i.e. constant flow and load conditions), and 1,0 are shown. Also indicated is the denitrification potential calculated from the empirical model Eq(5.18 or 5.19a).

From Fig 5.16 the denitrification potential is only marginally affected by the severity of cyclicity in the influent flow and load pattern. The reason for this insensitivity is due to the nature of the denitrification reaction, which is associated with utilization of organic substrate: The rate of utilization of easily biodegradable substrate is very high, and irrespective of the influent flow and load pattern the removal of this substrate in the pre-denitrification reactor, essentially,will be complete unless the pre-denitrification reactor is very small. The rate of utilization of slowly biodegradable material is not high, but in this case the unutilized material during peak loads is adsorbed on the sludge flocs and is utilized during periods of low load. Hence the utilization of biodegradable substrate is almost complete irrespective of the load pattern and, correspondingly, the associated denitrification potential is substantially unaffected by variations in flow or load. Although the denitrification potential is not affected by the influent flow and load pattern, the actual extent of nitrate removal may decrease due to the fact that the minimum nitrate concentration in the recycles may coincide with a maximum in the influent COD load. This may cause that the nitrate concentration in the anoxic zone becomes zero, that is, periodically the anoxic zone may be under-loaded with respect to nitrates. As a consequence, some of the substrate that could have been utilized in the anoxic zone is discharged into the subsequent aerobic zone where it is utilized and lost for purposes of

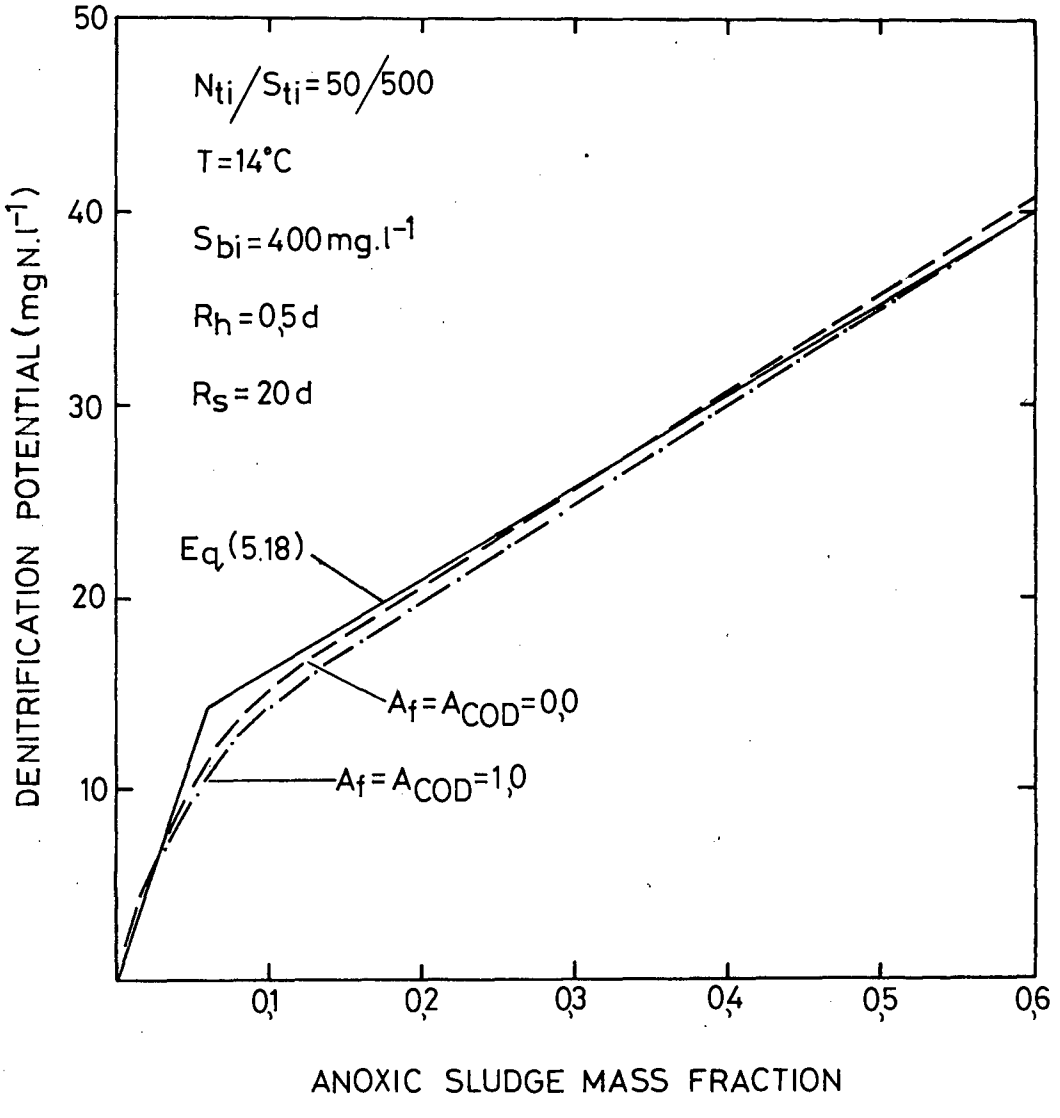

Fig. 5.16 *Denitrification potential of a predenitrification system for sionoidal input waves of flow and influent TKN and COD concentrations as a function of the anoxic sludge mass fraction.*

denitrification. In practice this implies that the required recycle ratio of a system under cyclic flow and load conditions must always be higher than of an equivalent system under constant flow and load conditions.

6.3 Nitrogen removal under cyclic flow and load conditions

When Figs 5.15 and 5.16 are compared it is evident that the reduction in nitrification capacity (reflected in an increase of the average effluent ammonia concentration) with increasing severity of cyclic input of flow and/or load is much more pronounced than that of the denitrification potential. As both the nitrification capacity and the denitrification potential decrease with increasing severity of the influent cyclic flow and load pattern, it can be concluded that the average concentration of the sum of the soluble nitrogen species in the effluent will increase with increasing severity of the influent cyclic flow and load pattern. This is illustrated in Fig 5.17 where the simulated values of the average total soluble effluent nitrogen concentration ($N_a + N_o + N_{ne}$) from a Modified Ludzack Ettinger process is shown as a function of the anoxic sludge mass fraction for different fractional amplitudes of sinusoidal input waves of flow, and influent TKN and COD concentrations (all in phase) for the same operational conditions as for the plots in Figs 5.15 and 5.16. Fractional amplitudes of 0,0 (i.e. constant flow and load), 0,5 and 1,0 are shown.

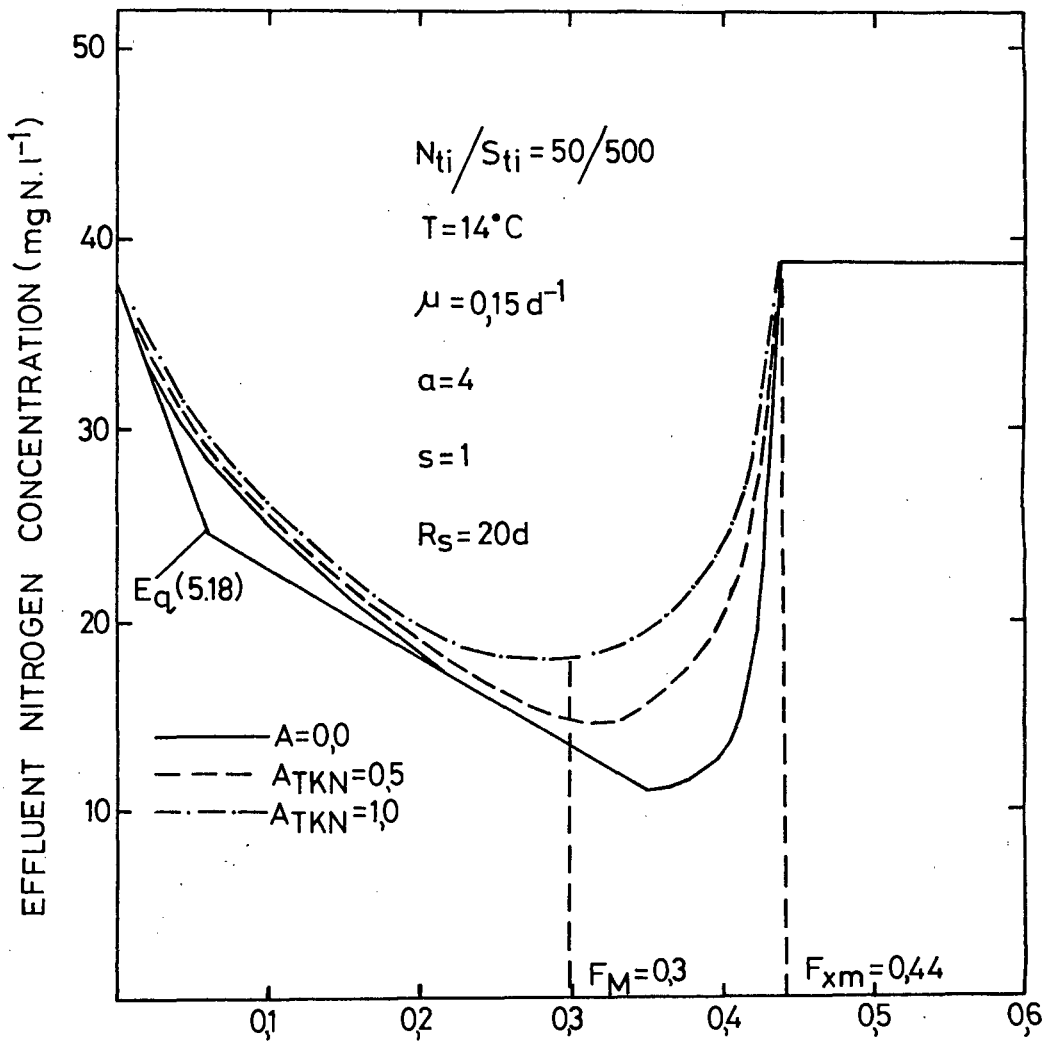

Fig. 5.17 *Mean total soluble nitrogen effluent concentration from a predenitrification system for sinoidal input waves of flow and influent TKN and COD concentrations as a function of the anoxic sludge mass fraction.*

The total soluble nitrogen effluent concentration based on the empirical model, Eq (5.18), is also shown. From Fig 5.17, while the nitrogen removal efficiency is reduced with increasing severity of the cyclic influent flow and load pattern this reduction is not very significant for fractional amplitudes to be expected in practice. (Fractional amplitudes of 1,0 are very unlikely to occur in a practical design situation). Furthermore the optimal value for the anoxic sludge mass fraction under constant flow and load conditions (f_M) calculated from Eq (5.13b) also gives optimal or near optimal results for cyclicly varying flow and load conditions. For these reasons the design procedure for constant flow and load conditions outlined in Section 5 of this chapter will provide a design that is also near optimal for cyclic flow and load conditions, although under the latter conditions the nitrogen removal efficiency is slightly less than under the former conditions.

7. DESIGN

In the previous sections of this chapter the factors that influence the nitrification and denitrification behaviour in single sludge systems were discussed in detail. Also it was shown that if these factors can be quantified, an optimal design for a nitrification-denitrification system can be obtained. The factors that influence the nitrification and denitrification behaviour can be categorised into two main groups:

(1) Sewage characteristics:

- (a) Influent flow and its variation over the day.
- (b) Influent COD concentration and its variation over the day.
- (c) Influent TKN concentration and its variation over the day.
- (d) Composition of the influent COD: f_{us}, f_{up} and f_{ca}.
- (e) Maximum specific growth rate of nitrifiers.
- (f) Denitrification rate constants.
- (g) Temperature range.

(2) Operational parameters:

(a) Recycle ratios 'a' and 's'

(b) Sludge age.

(c) Anoxic sludge mass fraction(s).

7.1 Sewage characteristics

In most design situations the design engineer will not have accurate or precise numerical values for the sewage characteristics. This will apply in particular when designing waste treatment plants for new areas; for extensions to existing plants the problem is perhaps less acute, as existing flows can be monitored to obtain all the relevant data. However, the instrumentation and capability to do this work, or the willingness to finance it is often lacking. In such circumstances the engineer is forced to make assumptions regarding sewage characteristics for which no quantitative estimate is available. Such assumptions will be of a conservative nature, so as to ensure that the system will work (i.e. it will nitrify and denitrify) though perhaps not optimally.

In the absence of data on the sewage characteristics the following considerations need to be taken into account when making assumptions for design:

(1) COD and TKN mass loading and influent TKN/COD ratio.

The daily COD mass loading will depend upon the size of the contributing population, the per capita contribution and the industrial contribution. The size of the population can be estimated from existing counts or projected future figures. The per capita contribution depends on the social, racial, and cultural characteristics of the population. In South Africa, for example, the per capita COD contribution for people of the white and coloured groups is approximately 100 to 110 g COD. capita^{-1}.d^{-1} whereas that of the Indian and African groups is

approximately 70 to 75 g COD.capita^{-1},d^{-1}. This lower figure will tend to merge with the higher as economic improvement occurs and cultural habits change with the adaption of an urban pattern of living.

A major qualitative difference also arises from public cleansing practice: If separate garbage collection is practised - as in South Africa - the figures for the per capita COD contribution above are probably operative; where garbage grinding and disposal to the sewer is practised - as often is the case in the U.S.A. - the per capita contribution could be as high as 200 g COD.day^{-1}.

The influent TKN/COD ratio, (N_{ti}/S_{ti},) can range from about 0,08 to 0,15 mg TKN.mg COD^{-1} for unsettled sewage. The "normal" ratio seems to be about 0,10 mg TKN.mg COD^{-1}. In general the TKN/COD ratio, after primary settling, tends to be greater than that for unsettled sewage. Another factor influencing the influent TKN/COD ratio is the retention time in the sewer. From measurements of the Strandfontein-Cape Town sewer, the TKN/COD ratio is well above 0,1 mg TKN.mg COD^{-1} apparently due to the long flat sewer line which retains the sewage up to 12 hours. Anaerobic destruction of the COD seems to take place whereas the TKN is largely unaffected. Hence the TKN/COD ratio tends to increase with increasing retention time in the sewer. Similarly septic tank discharge to sewers gives rise to high influent TKN/COD ratios. These examples are quoted as indicative of the wide range of sewage characteristics that can be encountered, of which the design engineer must take cognisance. Generally in South Africa there is a lack of knowledge concerning the relavent sewage characteristics of wastewaters for the design of wastewater treatment plants.

The per capita daily sewage flow suffers from the same lack of knowledge as the daily per capita COD and TKN contributions. Again economic, racial and associated cultural differences have a marked effect. Also the availability or non-availability of piped water (to standpoints or to each house) give rise to very different quantities of wastewater per capita per day.

(2) Composition of the influent COD.

The factors of importance in the influent COD, S_{ti}, are the unbiodegradable fractions, both soluble (f_{us}) and particulate (f_{up}) and the easily biodegradable fraction (f_{ca}) of the total biodegradable influent COD concentration (S_{bi}):

$$X_{ii} = f_{up}.S_{ti}$$

$$S_{ui} = f_{us}.S_{ti}$$

$$S_{bi} = (1-f_{us}-P.f_{up}).S_{ti}$$

$$S_{bs} = f_{ca}.S_{bi}$$

The fraction f_{up}, in South Africa, is approximately 0,09 for unsettled, and 0,02/0,05 mg VSS.mg COD^{-1} for settled sewage (depending on the efficiency of the primary settler). Where garbage grinding with disposal to the sewers is practised, the value of f_{up} seems to be high, up to 0,17 mg VSS.mg COD^{-1}; values for f_{up} for settled sewage, under these conditions, are not available. The fraction f_{ca} appears to be approximately 0,24 for both settled and unsettled sewage. This fraction, however, may be lower when the sewage is retained for long periods in the sewers - the indications are, for example, that the sewage from the Strandfontein sewer has an f_{ca} value of about 0,15, probably due to the long retention time (12 hours) in the sewer. The effect of industrial waste flows on f_{ca} is quite unknown.

(3) Kinetic constants of nitrification and denitrification

The maximum specific growth rate of nitrifiers μ_{nmT} can vary from 0,17 to 0,65 d^{-1} and appears to be correlated with the industrial fraction in the waste flow - the higher this fraction the lower the value of the constant. However, the value of μ_{nmT} is specific to each sewage source so that it would be injudicious to assign, for example, a high μ_{nmT} value based solely on the lack of an industrial waste fraction: If no experimental

determination is available, necessarily a low value for μ_{nmT} is indicated for purposes of design.

The denitrification rate constants K_2 and K_3 also are different for different waste flows but the variability of these constants is much smaller than that of the maximum specific growth rate of nitrifiers.

(4) Temperature

The temperature of the waste flow determines to a large degree the mixed liquor temperature in the activated sludge system. The sewage temperature is determined by the climatological conditions prevailing and the ground temperature at about 1m depth. The design engineer may benefit from experimental data obtained in plants already constructed and operating in nearby areas, in order to estimate the maximum and minimum temperatures to be expected. The critical aspects of the design will always be based on the minimum temperature to be expected; at any temperature higher than the minimum the performance of the plant will tend to improve.

The discussion above highlights the difficulties the design engineer has to resolve before embarking on a design. It is evident that the better the sewage characteristics are quantified, the more precise a design can be delineated and the more likely it becomes that the design will achieve true optimal nitrogen removal under the given conditions. Ideally laboratory and pilot scale studies should be carried out to define the sewage characteristics as described earlier in Chapter 4, but often such studies will not be possible for practical or economic reasons.

In the absence of adequate data, from the information on the behavioural characteristics of nitrification-denitrification plants, as set out in this report, conservative design requires that the estimate of the influent TKN/COD ratio should be on the high side, the f_{ca} fraction on the low side and the μ_{nmT} value on the low side at the minimum temperature to be expected. This will increase the chances that nitrification in the plant will be efficient under the

7.2 Operational Parameters

(1) Recycle ratios

From the theory of process optimization as set out in Section 5 of this Chapter, maximum nitrate removal is obtained for the highest total recycle (a+s), with the proviso that an increase of (a+s) is only meaningful if the nitrate available for denitrification in the pre-denitrification reactor is less than its denitrification potential. Considerations for selecting the total maximum recycle ratio are as follows:

(a) The underflow recycle, s, can range from 0,5 to 2 and is limited to this range by the requirements for effective functioning of the secondary settling tank. Low recycle ratios, of say, 0,5, increase the chances of zone settling failures particularly as experimental evidence on the settling characteristics indicates that the presence of the anoxic zone generally causes a deterioration of the zone settling characteristics of the mixed liquor relative to those from an aerobic 'system'. Also a low underflow recycle causes a build up of sludge in the settler so that denitrification is enhanced and causes floatation and rising sludge. High recycle ratios again may cause turbulance in the tank and lead to a poor effluent quality. An underflow recycle that appears to be satisfactory, judging from full scale behaviour, is $s = 1$. The difficulty at present is that very little is understood quantitatively of the secondary tank behaviour with respect to the zone settling properties of the sludge, so that the causes of malfunction of the tank often cannot be assigned: These causes may include inadequate design due to the settling properties of the sludge being very different from those implied when using "normal" design criteria, or inadequate operation due to a lack of understanding of the response of the settler to changing recycle ratios.

Another consideration is aeration of the pre-denitrification zone by dissolved oxygen in the underflow recycle. A common method for the underflow recycle is the Archimedean screw pump.

This pump serves as an efficient aerator of the underflow mixed liquor; dissolved oxygen concentrations of up to 5 mg.ℓ^{-1} are not uncommon after the flow has passed through the pump. The oxygen thus introduced into the pre-denitrification reactor reduces the denitrification capacity of this reactor. Although this reduction is not significant it does point towards

(i) keeping the 's'-recycle at relatively low values, and

(ii) keeping to the lift height of the pump and hence the opportunity for aeration as low as possible.

(b) the 'a'-recycle is from the aerobic reactor to the pre-denitrification reactor. Pumping is often by low lift (say 5-10 cm) propeller pumps. Very little oxygen is entrained by this pump; the oxygen source being almost wholly the oxygen present in the mixed liquor. At high recycle ratios the oxygen transferred to the anoxic reactor can be considerable if the dissolved oxygen concentration in the aerobic reactor is high. If the recycle channel is long, the dissolved oxygen concentration will be reduced during the passage through the channel. Although the dissolved oxygen concentration entering the anoxic reactor <u>per se</u> now may be low, in fact the channel forms part of the anoxic reactor so that the denitrification capacity is affected as if the oxygen entering the channel is discharged to the anoxic reactor. Increasing the recycle ratio above about a = 4 contributes relatively little to making extra nitrate available to the pre-denitrification reactor, but may reduce the denitrification capacity of this reactor by excessive introduction of dissolved oxygen. Hence from both a process and practical point of view recycle ratios higher than a = 4 are not merited. However, for flexibility of operation provision should be made for an 'a'-recycle of up to a = 6 to 8.

(2) Process configuration and sludge age.

In a particular design situation, diagrams similar to the one plotted in Fig 5.14 are very useful to determine the optimal process configuration and sludge age. From Figure 5.14, at any particular sludge age, there are three ranges of influent TKN/COD ratios, (N_{ti}/S_{ti}), defining an optimal process configuration:

(i) At low (N_{ti}/S_{ti}) ratios, $(N_{ti}/S_{ti}) < (N_{ti}/S_{ti})_o$, complete denitrification in a Bardenpho configuration is possible. In this configuration the recycle ratios 'a' and 's' are the maximum selected.

(ii) At medium (N_{ti}/S_{ti}) ratios, $(N_{ti}/S_{ti})_o < (N_{ti}/S_{ti}) < (N_{ti}/S_{ti})_1$ the system denitrification is incomplete; complete denitrification is possible only in the pre-denitrification reactor (again with maximum recycle ratios 'a' and 's'), in the post-denitrification reactor denitrification is incomplete.

(iii) At high (N_{ti}/S_{ti}) ratios, $(N_{ti}/S_{ti}) > (N_{ti}/S_{ti})_1$, the maximum nitrate removal takes place in a system without a post-denitrification reactor, i.e. a Modified Ludzack Ettinger system becomes the indicated system. At the maximum (a+s) recycle ratio the nitrate available to the pre-denitrification reactor is now greater than the denitrification potential - the total recycle ratio may be decreased to such a value that a complete denitrification in the pre-denitrification reactor just occufs.

To construct the diagram in Fig 5.14, information concerning all the sewage characteristics discussed in the previous section, together with maximum recycle ratios 'a' and 's' needs to be available either from experiment or by estimation. From Fig 5.14 option (ii) i.e., the Bardenpho configuration with incomplete denitrification lies within a narrow range of influent TKN/COD ratios. The data on the plant normally are unlikely to be precise to the degree the plant can be guaranteed to operate within such a narrow range even if the data indicates it should. Thus in practice one would tend to distinguish either a "low" TKN/COD ratio of the influent and design for complete denitrification (i.e., use a Bardenpho configuration) or a "high" TKN/COD ratio of the influent and design a pre-denitrification system (i.e., use a Modified Ludazck-Ettinger configuration). The division between these two categories is a subjective one; for TKN/COD ratios near the division line, the effluent nitrate concentration for either the Bardenpho or a Modified Ludzack Ettinger system are very similar.

Regarding the optimal sludge age of single sludge nitrification—denitrification systems, this value can also be determined with the aid

of a diagram such as in Fig 5.14. The extent of denitrification that can be obtained in either a Bardenpho or a Modified Ludzack Ettinger configuration increases considerably from the respective minimum sludge ages R_B and R_{MLE} until a sludge age R_{so} is attained, the sludge age R_{so} being the minimum sludge age that allows efficient nitrification at the largest anoxic sludge fraction f_{max} allowed (recommended f_{max} = 0,45 to 0,50). For a temperature of 14°C in the absence of experimental data concerning sewage characteristics the value of R_{so}, conservatively, is about R_{so} = 25 d (See Fig 5.12).

In practice the sludge age R_{so} must always be chosen for high TKN/COD ratios unless a lower sludge age is dictated by economic considerations concerning the cost of the plant. From Fig. 5.14 the maximum TKN/COD ratio that allows complete denitrification, $(N_{ti}/S_{ti})_o$ at a sludge age R_{so} is about 0,095 mg TKN.mg COD^{-1} for the conservative value of the maximum specific growth rate of nitrifiers: μ_{nm14} = 0,18 d^{-1}. Hence if the TKN/COD ratio in the influent (N_{ti}/S_{ti}) > 0,095 mg TKN/COD^{-1} complete denitrification is not possible and the sludge age R_{so} is the optimal sludge age of the system. However if the influent TKN/COD ratio (N_{ti}/S_{ti}) is considerably smaller than 0,095 mg TKN.mg COD^{-1} complete denitrification can be obtained at a sludge age $R_s < R_{so}$. For example, again using Fig 5.14, the required sludge age to obtain complete denitrification for an influent TKN/COD ratio of (N_{ti}/S_{ti}) = 0,08 mg TKN.mg COD^{-1} is only about 17 days, so that theoretically this sludge age is optimal. However, the sludge age R_s = 17d in this case might not be considered adequate for practical reasons: (1) A sludge age of 17 days will produce a sludge that is not sufficiently stabilized for direct discharge. (2) If a sludge age is chosen longer than R_s =17d, then the system has a capacity to remove all or more of the influent nitrates during periods when the TKN/COD ratios are higher than the estimated value of (N_{ti}/S_{ti}) = 0,08 mg TKN. mg COD^{-1}. The longer sludge age would also assure that efficient nitrification would take place even if the μ_{nm14} value would be lower than the estimated value (μ_{nm14} = 0,18 d^{-1}). Thus in the case of a low TKN/COD ratio in the influent, when theoretically complete denitrification could occur at a sludge age $R_s < R_{so}$, the design engineer is confronted with the choice of having to trade off the advantage of a low sludge age (i.e. a smaller, more economical plant) against the

advantage of a long sludge age, R_{so}, (i.e. good sludge stabilization, stability of operation and more consistently low effluent ammonia and nitrate concentrations.

7.3 Design example

The design example below serves to illustrate the optimization procedure in the case that all the relevant information concerning sewage characteristics is known.

Assuming the sewage characteristics and operational conditions listed below, design a nitrification denitrification system for the following influent TKN concentrations:

(a) $N_{ti} = 40$ mg N.ℓ^{-1}

(b) $N_{ti} = 50$ mg N.ℓ^{-1}

(c) $N_{ti} = 60$ mg N.ℓ^{-1}

Sewage characteristics

$S_{ti} = 500$ mg COD.ℓ^{-1}

$f_{us} = 0{,}07$ mg COD.mg COD^{-1}

$f_{up} = 0{,}09$ mg VSS.mg COD^{-1}

$f_{ca} = 0{,}24$ mg COD.mg COD^{-1}

$\mu_{nmT} = 0{,}26\ (1{,}123)^{T-20}\ d^{-1}$

$14^{o}C < T < 20^{o}C$

At $14^{o}C$: $K_2 = K_3 = 0{,}065$ mg N.mg Xa^{-1}.d^{-1}

At $20^{o}C$: $K_2 = 0{,}1$ and $K_3 = 0{,}079$ mg N.mg X_a^{-1}.d^{-1}

Operational conditions:

$R_s \leqslant 20d$

$S_f = 1,25$ at 14^oC

$s = 1$ (fixed)

$a = 4$ (maximum)

A) Estimate the maximum anoxic sludge mass fraction. In order to guarantee efficient nitrification throughout the year use $T = 14^oC$, the expected minimum temperature. For $T = 14^oC$ from Table 3.7 ($\mu_{nm20} = 0,36\ d^{-1}$).

$\mu_{nm14} = 0,18\ d^{-1}$

$bn_{14} = 0,034\ d^{-1}$

$K_{n14} = 0,5\ d^{-1}$

Hence for the maximum allowable sludge age $R_s = 20$ d and the imposed safety factor for nitrification $S_f = 1,25$: (Eq (5.13b)).

$$f_M = 1 - S_f(b_{n14} + 1/R_s)/\mu_{nm14}$$

$$= 0,42$$

B) Estimate the ratio (nitrification capacity / biodegradable influent COD), (N_c/S_{bi})

1) At 14^oC and for $R_s = 20d$ using Eq (5.16):

$$N_s = f_n\ (Y_h(1+f\ b_{hT}R_s)(1-f_{us}-P.f_{up})/(1+b_{hT}R_s)+f_{up})\ S_{ti}$$

$$= 11\ mg\ N.\ell^{-1}$$

$N_o = 1\ mg\ N.\ell^{-1}$ (estimated)

$N_a = 2\ mg\ N.\ell^{-1}$ (given)

Table 5.1 Nitrification capacity and N_c/S_{bi} ratios for the different influent TKN concentrations at the maximum and minimum temperature

		T = 14°C		T = 20°C	
N_{ti}	N_{ti}/S_{ti}	N_c	N_c/S_{bi}	N_c	N_c/S_{bi}
40,0	0,08	26,0	0,065	27,6	0,069
50,0	0,10	36,0	0,090	37,6	0,094
60,0	0,125	46,0	0,115	57,6	0,119

C) From Eq (5.38) the maximum ratio N_c/S_{bi} that allows complete denitrification for maximum values of recycles (a=4 and s=1) sludge age R_s=20d and anoxic sludge mass fraction (f_M=0,42) i.e. $(N_c/S_{bi})_o$, can be calculated for the maximum and minimum temperature:

At 14°C: $(N_c/S_{bi})_o$ = 0,077 mg N.mg COD^{-1}

At 20°C: $(N_c/S_{bi})_o$ = 0,086 mg N.mg COD^{-1}

Similarly the maximum (N_c/S_{bi}) ratio for which it is advantageous to have a post-denitrification reactor for the given conditions, $(N_c/S_{bi})_1$, is calculated using Eq (5.47):

At 14°C: $(N_c/S_{bi})_1$ = 0,093 mg N.mg COD^{-1}

At $20^{o}C$: $(N_c/S_{bi})_1 = 0{,}112$ mg N.mg COD^{-1}

D) $N_{ti} = 40$ mg $N.\ell^{-1}$

For $N_{ti} = 40$ mg $N.\ell^{-1}$ the calculated ratios (Nitrification capacity/ biodegradableinfluent concentration) at $14^{o}C$ and at $20^{o}C$ are lower than the respective values of the maximum $(N_c/S_{bi})_o$ ratios that allow complete denitrification:

At $14^{o}C$: $(N_c/S_{bi}) = 0{,}065 < (N_c/S_{bi})_o = 0{,}077$

At $20^{o}C$: $(N_c/S_{bi}) = 0{,}069 < (N_c/S_{bi})_o = 0{,}086$

Hence for $N_{ti} = 40$ mg $N.\ell^{-1}$ complete nitrate removal is possible both at $14^{o}C$ and at $20^{o}C$. It is possible even to decrease the sludge age: Using Fig 5.14, the minimum sludge age that allows complete denitrification is about 17 d.

Check: for $R_s = 17$ d using Eq (5.38):

At $14^{o}C$: $(N_c/S_{bi})_o = 0{,}067 > (N_c/S_{bi}) = 0{,}065$

At $20^{o}C$: $(N_c/S_{bi})_o = 0{,}081 > (N_c/S_{bi}) = 0{,}069$

For $R_s = 17$ d the maximum anoxic sludge mass fraction at $14^{o}C$ is given by Eq (5.13):

$f_M = 0{,}35$. For a = 4 and s = 1 the mass sludge fraction in the pre-denitrification reactor must be Eq (5.34): $f_{x1} = 0{,}14$ and consequently $f_{x3} = f_M - f_{x1} = 0{,}35 - 0{,}14 = 0{,}21$.

Thus the "optimal" process for $N_{ti} = 40$ mg $N.\ell^{-1}$ is characterised by the following parameters:

System configuration:	Bardenpho
Anoxic sludge mass fraction:	$f_M = 0,35$
of which pre-denitrification:	$f_{x1} = 0,14$
post-denitrification:	$f_{x3} = 0,21$
Sludge age:	$R_s = 17d$
Recycle ratios:	$a = 4$
	$s = 1$

Under these conditions denitrification is complete and the total soluble nitrogen effluent concentration, N_t, is composed of residual organic and ammoniacal nitrogen only:

At 14^oC: $N_t = N_o+N_a = 1,0 + 2,0 = 3,0$ mg $N.\ell^{-1}$

At 20^oC: $N_t = N_o+N_a = 1,0 + 0,7 = 1,7$ mg $N.\ell^{-1}$

At 20^oC the denitrification potential exceeds the nitrification capacity: $D_{p1} = (\alpha+K_2C_rf_{x1})S_{bi} = 19,7$; $D_{p3} = K_3C_rf_{x3}S_{bi} = 10,0$ so that $D_p = D_{p1}+D_{p3} = 29,7$ mg $N.\ell^{-1}$ whereas $N_c = 27,6$ mg $N.\ell^{-1}$. The <u>minimum</u> recycle required at 20^oC can be calculated as follows: The denitrification potential of the post-denitrification reactor $D_{p3} = 10$ mg $N.\ell^{-1}$ so that in the pre-denitrification reactor the nitrate removal must be : $N_c-D_{p3} = 27,6 - 10,0 = 17,6$. Hence $N_{av1} = a/(a+s+1).N_c = 17,6$ and $a=3,5$.

Instead of reducing the sludge age from $R_s = 20$ to $R_s = 17$ d it is also possible to keep $R_s = 20$ d and to reduce the anoxic sludge mass fractions, because for $R_s = 20$ d the anoxic sludge mass fraction need not be the maximum allowable ($f_M = 0,42$) to achieve complete denitrification. Using Eqs (5.34) and (5.36) the minimum anoxic sludge mass fractions for complete denitrification can be calculated: At 14^oC:

$$f_{x1} = (a/(a+s+1)\ (N_c/S_{bi})\ -\alpha)/(K_2C_r) = 0,13$$

$$f_{x3} = (s+1)/(a+s+1)(N_c/S_{bi})/(K_3C_r) = 0,19$$

Hence for $R_s = 20$ d it is possible to operate the system with a

total anoxic sludge mass fraction of $f_x = f_{x1} + f_{x3} = 0,13 + 0,19 = 0,32$, smaller than the maximum allowable $f_M = 0,42$. It can be verified that $f_{x1} = 0,13$ and $f_{x3} = 0,19$ are sufficient for complete denitrification at 20°C.

$$D_{p1} = (\alpha + K_2 C_r f_{x1}) S_{bi} = 19,4 > N_c.a/(a+s+1) = 17,3$$

$$D_{p3} = K_3 C_r f_{x3} S_{bi} = 11,9 > N_c(s+1)/(a+s+1) = 8,7$$

E) $N_{ti} = 50$ mg $N.\ell^{-1}$

For $N_{ti} = 50$ mg $N.\ell^{-1}$ the calculated ratios (Nitrification capacity/biodegradable influent concentration) at 14°C and at 20°C exceed the respective values of $(N_c/S_{bi})_o$ but are lower than the $(N_c/S_{bi})_1$ values:

At 14°C: $(N_c/S_{bi})_o < (N_c/S_{bi}) < (N_c/S_{bi})_1$

$(0,077 < 0,090 < 0,093)$

At 20°C: $(N_c/S_{bi})_o < (N_c/S_{bi}) < (N_c/S_{bi})_1$

$(0,086) < 0,094 < 0,0112)$

Hence complete denitrification is not possible but a post-denitrification reactor is advantageous. The division of the maximum anoxic sludge mass fraction over pre- and post-denitrification reactor can now be established as follows:

Using Eq (5.45):

At 14°C : $f_{x1} = 0,38$

At 20°C: $f_{x1} = 0,26$

To decide which anoxic sludge fraction to choose in this case it must be remembered that at 14°C the denitrification rate constants

K_2 and K_3 are equal and that the division of the anoxic sludge mass fraction f_M over the pre- and post-denitrification reactors does not affect the system nitrate removal while $f_{x1} > f_{min}$. Hence the value $f_{x1} = 0,26$, which is the optimal value for T = 20°C, is chosen. For $f_{x1} = 0,26$ the anoxic sludge mass fraction in the post-denitrification reactor is:

$$f_{x3} = f_M - f_{x1} = 0,42 - 0,26 = 0,16$$

The nitrate removal performance can now be calculated:

(a) At 20°C:

$$D_{p1} = (\alpha + K_2 C_r f_{x1}) S_{bi} = 27,4 \text{ mg N.}\ell^{-1}$$

$$D_{p3} = K_3 C_r f_{x3} S_{bi} = 7,9 \text{ mg N.}\ell^{-1}$$

Hence the effluent nitrate concentration

$$N_{ne} = N_c - D_{p1} - D_{p3} = 37,6 - 27,4 - 7,9 = 2,3 \text{ mg N.}\ell^{-1}$$

(Check:

$$N_{av1} = a/(a+s+1)\ N_c + sN_{ne} = 27,4 = D_{p1}$$

i.e., the nitrate available in the pre-denitriciation reactor for maximum'a'and's'recycles is equal to the denitrification potential)

(b) At 14°C:

Obviously for $f_{x1} = 0,26$ the 'a' recycle does not have to be a=4 in order to convey sufficient nitrate to the pre-denitrification reactor.

The optimal recycle using Eq (5.44)

$$D_{p1} = (a+s)/(a+s+1).N_c - s/(s+1)D_{p3}$$

$$D_{p1} = \alpha + K_3 C_r f_{x1} = 23,4 \text{ mg N.}\ell^{-1}$$

$$D_{p3} = K_3 C_r f_{x3} S_{bi} = 7,5 \text{ mg N.}\ell^{-1}$$

i.e., $a + s = 3,1$

$a = 2,1$

$s = 1,0$

The effluent nitrate concentration N_{ne} is given by:

$$N_{ne} = N_c - D_{p1} - D_{p3} = 36 - 23,4 - 7,5 = 5,1 \text{ mg N.}\ell^{-1}$$

Check with Eq (5.42):

$$N_{avl} = a.N_c/(a+s+1) + sN_{ne} = 2,1.36/4,1 + 5,1$$

$$= 23,4$$

$$= D_{p1}$$

The minimim total soluble effluent nitrogen concentration can now be calculated:

At 14°C: $N_t = N_{ne} + N_a + N_o = 5,1 + 2,0 + 1,0 = 8,1 \text{ mg N.}\ell^{-1}$

At 20°C: $N_t = N_{ne} + N_a + N_o = 2,3 + 0,7 + 1,0 = 4,0 \text{ mg N.}\ell^{-1}$

F) $N_{ti} = 60 \text{ mg N.}\ell^{-1}$

For $N_{ti} = 60 \text{ mg N.}\ell^{-1}$ the calculated ratios (Nitrification capacity/biodegradable influent concentration) at 14°C and at 20°C exceed the respective values of $(N_c/S_{bi})_1$:

At 14°C: $(N_c/S_{bi}) = 0,115 > (N_c/S_{bi})_1 = 0,093$

At 20°C: $(N_c/S_{bi}) = 0,119 > (N_c/S_{bi})_1 = 0,112$

Hence for this influent TKN concentration the Modified Ludzack Ettinger configuration is the optimal system. The effluent nitrate concentration are easily calculated from the differences between N_c and D_{p1}.

At 14°C:

$$N_{ne} = N_c - D_{p1}$$

$$= 46,0 - (\alpha + K_2 C_r f_{x1}) S_{bi}$$

$$= 46,0 - 30,9 = 15,1 \text{ mg N.}\ell^{-1}$$

At 20°C:

$$N_{ne} = N_c - D_{p1}$$

$$= 47,6 - 37,5 = 10,1 \text{ mg N.}\ell^{-1}$$

The required 'a' recycles to introduce sufficient nitrate into the anoxic reactor is calculated from $N_{av1} = D_{p1}$ i.e.

at 14°C: $D_{p1} = 30,9 = N_{av1} = 46,0\ (a+s)/(a+s+1)$ i.e. $a = 1,0$

at 20°C: $D_{p1} = 37,5 = N_{av1} = 47,6\ (a+s)/(a+s+1)$ i.e. $a = 2,7$

In Table 5.2 are listed the most important characteristics of the process configuration and operational conditions for the different influent TKN concentrations and at the minimum and maximum temperatures.

Table 5.2 Optimal reactor configuration effluent quality and recycle ratio for different N_{ti}/S_{ti} ratios at minimum and maximum temperatures

	N_{ti} = 40 mg N.ℓ^{-1}		N_{ti} = 50 mg N.ℓ^{-1}		N_{ti} = 60 mg N.ℓ^{-1}	
System	Bardenpho		Bardenpho		MLE	
$R_{s(min)}$	17 d		20 d		20 d	
f_M	0,35		0,42		0,42	
f_{x1}	0,14		0,26		0,42	
f_{x3}	0,21		0,16		-	
	T=14°C	T=20°C	T=14°C	T=20°C	T=14°C	T=20°C
N_c	26,0	27,6	36,0	37,6	46,0	47,6
D_{p1}	17,5	19,7	23,4	27,4	30,9	37,5
D_{p3}	8,6	10,0	7,5	7,9	-	-
D_p	26,0	27,6	30,9	35,3	30,9	37,5
N_{ne}	0,0	0,0	5,1	2,3	15,1	10,1
N_a	2,0	0,7	2,0	0,7	2,0	0,7
N_t	3,0	1,7	8,1	4,0	18,1	10,8
a	4,0**	3,5	2,1	4,0**	1,0	2,7

** maximum 'a'-recycle ratio

CHAPTER SIX

CONCLUSIONS

This investigation encompassed a wide-ranging and in-depth enquiry into the behaviour of the single sludge nitrification-denitrification activated sludge process. The principal conclusions can be summarized as follows :

1. GENERAL PROCESS KINETICS

The bisubstrate-active site-death regeneration model developed by Dold, Ekama and Marais (1980) to describe the aerobic activated sludge process was extended to provide a reliable description of the kinetic behaviour of the single sludge nitrification-denitrification process. It was found that the kinetics of the activated sludge process in an anoxic environment can be described by the same basic equations that describe the kinetics of the activated sludge process in an aerobic environment, provided cognisance is taken of the following restrictions :

(1) To describe quantitatively the rate of denitrification and the associated rate of substrate utilization in an anoxic environment, it is necessary to change the value of only one kinetic constant namely the specific utilization rate constant for slowly biodegradable material in an anoxic environment, K_{mp}^{1}. The value of K_{mp}^{1} was found to be a fraction of 0,38 of the value of the specific utilization rate constant for slowly biodegradable material in an aerobic environment, K_{mp}.

(2) Growth of the nitrifiers takes place only in an aerobic environment but death of the nitrifiers is assumed to take place in both the anoxic and the aerobic environment at the same rate as in an aerobic environment.

By accepting the above two restrictions, it was possible to predict accurately the response of nitrification-denitrification plants under different operational conditions(temperatures from 14 to 20°C; sludge ages from 6 to 20 days), process configurations(pre- and post-denitrification reactor; plug flow and completely mixed reactors), operational modes (continuous anoxic reactors and alternatingly anoxic-aerobic reactors) and influent flow and load conditions(constant and cyclically varying flow and load).

A particularly useful extension to the general model was the inclusion of a description of the kinetics of the mixed liquor alkalinity. This was done by assuming that alkalinity changes in the activated sludge process are due only to the effects of nitrification, denitrification, ammonification and de-ammonification. The predicted and observed alkalinity-time behaviour correlated so well that the alkalinity served as a reliable parameter additional to the parameters of nitrogenous species (ammonia, TKN and nitrate)to analyse nitrification and/or denitrification kinetics. It was found also that alkalinity was useful when describing the behaviour of low alkalinity influents; when the alkalinity was predicted to decrease below 35 ppm $CaCo_3$, it was theoretically predicted and experimentally observed that the pH of the mixed liquor became unstable and decreased to values significantly lower than 7.

2. DENITRIFICATION KINETICS

2.1. Denitrification under Constant Flow and Load Conditions

The traditional model for denitrification describes the system nitrate removal in anoxic reactors under constant flow and load conditions as a zero order process :

$$\Delta N_s = K X_v R \qquad (6.1)$$

where

ΔN_s = system nitrate removal (mgN.ℓ^{-1})

X_v = volatile solids concentration (mgVSS.ℓ^{-1})

R = nominal retention time in the anoxic reactor(s)(h)

K = empirical denitrification rate constant (mgN.mgVSS^{-1}.h^{-1})

The disadvantage of this model is that the denitrification rate constant, K, is empiric, not linked with basic activated sludge theory. Consequently, it is difficult, indeed impossible, to circumscribe the condition necessary for, and the range of, its applicability. From experimental data it soon became evident that the expression for ΔN_s led to inconsistent results. It could be made more consistent by linking the system nitrate removal to the active sludge concentration, X_a :

$$\Delta N_s = K' X_a R \qquad (6.2)$$

where

X_a = active sludge concentration (mg VSS.ℓ^{-1}).

In order to estimate X_a it was assumed that the formulation for aerobic systems by Ekama and Marais (1976) was applicable also to anoxic/aerobic systems. This assumption appeared reasonable as it led to consistent predictions of sludge production over a wide range of sludge ages (3 to 35 days) and temperatures (7 to 25°C).

It was established experimentally that the denitrification rate "constant" K', indeed remained constant, and apparently independent of the sludge age, for sludge ages R_s>10d. However, from experimental nitrate concentration-time profiles in anoxic plug flow reactors, three different denitrification rate constants, K_1', K_2 and K_3 were identified, depending on the position of the plug flow reactor in the series, i.e. pre- or post-denitrification reactor.

(i) Predenitrification plug flow reactor.

In this reactor two phases could be distinguished :

(a) a primary phase with a high denitrification rate, associated with K_1', and

(b) a secondary phase with a lower denitrification rate, associated with K_2. Both rates were approximately constant and could be described as follows :

primary phase : $\Delta N/\Delta t = K_1' X_a$

secondary phase : $\Delta N/\Delta t = K_2 X_a$

(ii) Postdenitrification plug flow reactor.

In this reactor a single phase with an approximately constant rate was observed :

$\Delta N/\Delta t = K_3 X_a$

To establish a link between the empiric formulation and basic activated sludge theory, the general model was used to simulate the denitrification behaviour in anoxic plug flow reactors. The simulated response exhibited closely the same zero order behaviour observed experimentally. From this it was concluded that the apparent zero order characteristic was a consequence of the kinetic response of the process to the influent characteristics when the system was operated under constant flow and load conditions. This was established as follows :

(i) Predenitrification reactor.

In the primary phase of the predenitrification reactor the high rate of denitrification was due to utilization of <u>both</u> easily and slowly biodegradable substrate. In the general model the rates of denitrification due to utilization of these substrates are respectively :

$$dN_1/dt = -[((1-PY_h)/2,86).\ K_{msT}.S_{bs}/(S_{bs}+K_{ssT})]X_a \quad (6.3)$$

$$dN_2/dt = -[((1-PY_h)/2,86).\ K^1_{mpT}\ X_sP/(X_sP+K_{spT}X_a)]X_a \quad (6.4)$$

where

(dN_1/dt) = denitrification rate due to utilization of easily biodegradable material (mg $N.\ell^{-1}.d^{-1}$)

(dN_2/dt) = denitrification rate due to utilization of slowly biodegradable material (mg $N.\ell^{-1}d^{-1}$)

K_{msT} = specific utilization rate constant for easily biodegradable material (mg COD.mg $X_a^{-1}.d^{-1}$)

K^1_{mpT} = specific utilization rate constant for slowly biodegradable material (mg COD.mg $X_a^{-1}.d^{-1}$)

K_{ssT} = half saturation value for easily biodegradable material (mg $COD.\ell^{-1}$)

K_{spT} = half saturation value for slowly biodegradable material (mg COD.mg X_a^{-1})

P = COD/VSS ratio (mg COD.mg VSS^{-1})

Y_h = Yield coefficient (mg VSS.mg COD^{-1})

X_s = concentration of stored material (mgVSS.ℓ^{-1}).

From simulations using plug flow reactors both (dN_1/dt) and (dN_2/dt) are approximately constant :

the denitrification rate due to utilization of easily biodegradable material, (dN_1/dt), is constant because the value of K_{ssT} is small and the Monod ratio $(S_{bs}/(S_{bs}+K_{ssT})$ deviates significantly from unity only when the concentration of easily biodegradable material becomes very small. Hence, while the S_{bs} concentration remains above a certain minimum value (about 10 mg $COD.\ell^{-1}$) the following equation applies very accurately :

$$(dN_1/dt) = -K_1 X_a \qquad (6.5)$$

where

K_1 = denitrification rate constant due to the utilization of easily biodegradable material.

The denitrification rate due to slowly biodegradable material, (dN_2/dt), in a predenitrification plug flow reactor, is approximately constant because the rate of substrate utilization is so small that the concentration of stored material, X_s, does not change significantly with retention time, i.e. the fraction $X_s P/(X_s P+K_{spT}X_a)$ in Eq. (6.4) is essentially constant and hence

$$(dN_2/dt) = -K_2 X_a \qquad (6.6)$$

During the primary phase, because the theoretical rate constants are independent of each other, the denitrification rate can be expressed as :

$$(dN/dt)_{prim} = -(K_1+K_2)X_a \qquad (6.7)$$

i.e. in terms of experimentally observed expression :

$$K_1' = K_1 + K_2.$$

Consequently the extent of nitrate removal in a plug flow denitrification reactor is :

$$\Delta N_{1s} = K_1 X_a R_{min} + K_2 X_a R_1 \qquad (6.8)$$

where

ΔN_{1s} = system nitrate removal in a predenitrification reactor ($mgN.\ell^{-1}$)

R_{min} = minimum nominal retention time for complete utilization of easily biodegradable material(h)

R_1 = nominal retention time in the predenitrification plug flow reactor(h).

Usually the value of K_1 is of little importance because in practical processes the retention time in the anoxic reactor is much longer than that required for complete utilization of the easily biodegradable influent material, S_{bsi}. When complete utilization of S_{bsi} takes place, the <u>extent</u> of nitrate removal becomes the parameter of interest, and is given by :

$$\begin{aligned} \Delta N'_{1s} &= (1 - PY_h)\ S_{bsi}/2{,}86 \\ &= (1 - PY_h)\ f_{ca}.\ S_{bi}/2{,}86 \\ &= \alpha.S_{bi}. \end{aligned} \qquad (6.9)$$

where

$\Delta N'_{1s}$ = system nitrate removal due to utilization of easily biodegradable material($mgNO_3$-$N.\ell^{-1}$)

S_{bi} = influent biodegradable COD concentration ($mgCOD.\ell^{-1}$)

S_{bsi} = influent easily biodegradable COD concentration ($mgCOD.\ell^{-1}$)

f_{ca} = S_{bsi}/S_{bi}

P = ratio COD/VSS for active sludge($mgCOD.mgVSS^{-1}$)

Y_h = yield coefficient for heterotrophic sludge growth($mgVSS.mgCOD^{-1}$)

α = proportionality constant($mgN.mgCOD^{-1}$)

Consequently the system nitrate removal in a predenitrification reactor with complete utilization of easily biodegradable material is given by :

$$\Delta N_{1s} = \alpha S_{bi} + K_2 X_a R_1 \qquad (6.10)$$

(ii) Postdenitrification reactor.

In a post-denitrification reactor, denitrification is due only to utilization of slowly biodegradable material. In this reactor, the concentration of stored material remains approximately constant and at a low value, due to the internal generation of slowly biodegradable material from death and lysis of organisms. Consequently, the rate of substrate utilization is constant and at a low value. Because of this behaviour again it is possible to describe denitrification closely by a zero order reaction :

$$dN_2/dt = -[((1-PY_h)/2,86)K^1_{mpT}X_sP/(X_sP+K_{spT}X_a)]X_a$$

$$= K_3 X_a \qquad (6.11)$$

and

$$\Delta N_{3s} = K_3 X_a R_3 \qquad (6.12)$$

where

ΔN_{3s} = system nitrate removal in a post-denitrification reactor ($mgN.\ell^{-1}$)

R_3 = nominal retention time in the post-denitrification reactor (h).

The value of K_3 is less than that of K_2 because the ratio $X_sP/(X_sP + K_{spT}X_a)$ in a post-denitrification reactor is smaller than that in a comparable pre-denitrification reactor.

From the work briefly described above the zero order denitrification behaviour in pre- and post-denitrification reactors can be explained adequately in terms of the general theory. The empirical denitrification rate constants in the zero order equations do not have a fundamental kinetic significance; they are apparent constants, the result of a combination of kinetic reactions that, fortuitously, show little variation in values for sludge ages longer than 10 days. Being empirical, their range of applicability is limited; from simulations it was found that,as the sludge age decreased below 10 days, the K values no longer remained "constant" but diverged markedly from the values for sludge ages longer than 10 days. This, however, does not constitute a significant limitation on the use of the empirical constants for design, because in nitrification-denitrification systems it is most unlikely that the operational sludge age will be less than 10 days.

2.2. Denitrification under cyclic flow and load conditions

Under cyclic flow and load conditions the rate of denitrification varies with time, being high during periods of high load(high substrate concentration) and low during periods of low load(low substrate concentration). However, from simulation using the general program, it was concluded that the average rate of denitrification and the mass of nitrate removed per day in an anoxic reactor under cyclic flow and load conditions, is almost identical to those under equivalent constant flow and load conditions. This conclusion is particularly important for design purposes; it allows the primarily design to be made and its average response to be evaluated without the need to take the cyclic flow and load into account.

3. NITRIFICATION KINETICS

3.1. Nitrification under constant flow and load conditions

In aerobic activated sludge systems under constant flow and load conditions, the minimum sludge age for nitrification used to be given by :

$$R_{sm} = 1/\mu_{nmT} \qquad (6.13)$$

where

R_{sm} = minimum sludge age for nitrification (d)

μ_{nmT} = maximum specific growth rate of nitrifiers (d^{-1})

Theoretically, taking cognisance of the death rate of nitrifiers, a superior expression is :

$$R_{sm} = 1/(\mu_{nmT} - b_{nT}) \qquad (6.14)$$

where

b_{nT} = death rate constant for nitrifiers (d^{-1})

When anoxic reactors are present in an activated sludge system, the minimum sludge age for nitrification must be increased such that the nitrifiers can maintain themselves in the aerobic reactors of the system.

From this experimental investigation it was concluded that, provided the actual time the nitrifiers are in an anoxic environment is not longer than 5 hours, the maximum specific growth rate of nitrifiers, μ_{nmT}, in an aerobic environment is not affected by the presence of anoxic reactors; this conclusion probably is valid only for this investigation. Past experience has shown that adverse effects may be experienced if the actual anoxic time exceeds 1½ hours.

From a theoretical analysis and from simulations using the general model, it was concluded that the minimum sludge age for nitrification in anoxic/aerobic systems is given by :

$$R_{sm} = 1/[\mu_{nmT}(1-f_x) - b_{nT}] \qquad (6.15)$$

where

$(1-f_x)$ = aerobic sludge mass fraction.

That is, the presence of anoxic reactors has an equivalent effect of reducing μ_{nmT} :

$$\mu'_{nmT} = (1-f_x)\mu_{nmT}$$

where

μ'_{nmT} = apparent maximum specific growth rate in anoxic/aerobic systems, (d^{-1})

The expression above, (Eq.6.15), for obtaining the minimum sludge age for nitrification in anoxic/aerobic systems, replaces the popular expression :

$$R_{sm} = 1/[(1-f_x)\mu_{nmT}] \qquad (6.16)$$

for estimating the minimum aerobic sludge age. Comparing Eqs.(6.15 and 6.16), utilization of Eq.(6.15) results in significantly larger values for the minimum sludge age, particularly at low temperatures.

Equation (6.15) above, refers to the minimum sludge age for nitrification to commence. One objective in nitrification-denitrification systems is to ensure efficient nitrification. Hence in practical nitrification-denitrification systems the operational sludge age should always be longer than the minimum, so that a high degree of nitrification is guaranteed. Using simulation results it would appear that the minimum sludge age for efficient nitrification R'_{sm}, should be longer, at least, by a factor 1,25 i.e.

$$R'_{sm} = 1,25\ R_{sm}.$$

If this rule is followed and the minimum sludge age is determined at the lowest temperature expected, using the lowest expected value for μ_{nmT} at the minimum temperature, then a nitrification efficiency of

more than 95% can be expected.

3.2. Nitrification under cyclic flow and load conditions

Simulation of nitrification response under cyclically varying flow and load conditions indicated that the nitrification efficiency was always lower than under equivalent constant flow and load conditions. At any selected sludge age, the reduction in efficiency was directly related to the intensity of the cyclicity of the influent TKN concentration and the flow. It was found possible to relate, approximately, the efficiency under cyclic conditions to that under average flow and load conditions as follows :

$$N_{a,cy} = N_{a,co} (1 + (A_{TKN} + A_f)/R_h) \qquad (6.17)$$

where

$N_{a,cy}$ = ammonia effluent concentration under cyclic flow and load conditions (mg N.ℓ^{-1})

$N_{a,co}$ = ammonia effluent concentration under constant flow and load conditions (mg N.ℓ^{-1})

A_{TKN} = fractional amplitude of the influent TKN concentration

A_f = fractional amplitude of the influent flow

R_h = total hydraulic retention time (d)

The causes for the reduction in nitrification efficiency can be explained as follows : During high flow or load periods, even though the organisms operate at maximum rate, it is not possible to oxidize all the influent ammonia and hence ammonia is discharged with the effluent. This in turn reduces the mass of nitrifiers formed in the system, that is, cyclicity, inter alia, has an effect equivalent to reducing the aerobic sludge age. For design, the reduction in efficiency can be countered always by ensuring that the sludge age is well above the minimum for nitrification. The rule, previously set

out, that the sludge age should not be less than 1,25 the minimum sludge age for nitrification, will ensure that nitrification is essentially complete even under cyclically varying flow and load conditions.

4. QUANTITATIVE VALUES OF CONSTANTS FOR NITRIFICATION AND DENITRIFICATION KINETICS

From calibration of the general model it became apparent that the kinetic constants in the differential equations describing nitrification and denitrification are affected in various degrees by influent sewage sources.

In nitrification, the pivotal parameter is the maximum specific growth rate of nitrifiers, μ_{nmT}. Usually the value of this constant is reported with respect to the reference temperature 20°C. It was found that μ_{nm20} can range from 0,17 to 0,65d^{-1} and is specific to a particular sewage source. Generally, the higher the industrial content of the wastewater, the lower the value of μ_{nm20}. The value of μ_{nm20} is so sensitive to the sewage source that variation in μ_{nm20} was observed even between batches of sewage from the same source. For this reason it was concluded that μ_{nm20} should be considered as a characteristic of the sewage, preferably to be determined for each sewage source.

The need for evaluating μ_{nm20} for each sewage is particularly evident when designing a nitrification-denitrification process for optimum nitrate removal. If μ_{nm20} is low, the aerobic sludge mass fraction necessarily must be higher than if μ_{nm20} is high and, consequently, the extent of denitrification achievable is lower for a low than for a high μ_{nm20} value. This was the reason for enquiring into ways and means for estimating μ_{nm20} rapidly and accurately.

In denitrification the fraction of influent easily biodegradable material, characterized by the mass parameter $f_{ca} = S_{bsi}/S_{bi}$ and the kinetic constant K^1_{mpT} were found to be pivotal constants that controlled the denitrification rate. The experimental evidence indicated that these constants were also affected by the sewage source; f_{ca} ranged between 0,05 and 0,24 and K^1_{mp20} between 0,66 and 1,14 mgCOD. mg X_a^{-1}.d.$^{-1}$. The values of the empirical parameters α, K_2 and K_3 were concomittantly affected, but the variability of K_2 and K_3 with varying K^1_{mpT} values is small, due to a compensatory effect in the expression for the utilization of slowly biodegradable material (X_s increases as K^1_{mp} decreases). For this reason it is believed that the "standard" values for the empirical constants, $K_2=0,042(1,08)^{T-20}$ and $K_3=0,0032(1,03)^{T-20}$ mgN.mgX_a^{-1}.h^{-1}, are adequate for design purposes.

The proportionality constant α (determined by the fraction of easily biodegradable material in the influent f_{ca}) was found to vary significantly between different sewage sources : values between 0,006 and 0,028 mg N.mg COD^{-1} (corresponding to f_{ca} values between 0,05 and 0,24) were measured for different wastewaters. The f_{ca} value appeared to be dependent also on the retention time in the sewer : long retention times may allow some anaerobic degradation to take place, the degradation reducing the easily biodegradable fraction disproportionately compared to the slowly biodegradable fraction of the sewage.

5. EXPERIMENTAL DETERMINATION OF KINETIC CONSTANTS FOR NITRIFICATION AND DENITRIFICATION

It was stated in Section 4 above that the fundamental constant that determines the rate of nitrification is the maximum growth rate of nitrifiers, μ_{nmT}; the fundamental constants that determine the rate of denitrification are : (i) the fraction easily biodegradable : total biodegradable influent COD, f_{ca}, and (ii) the utilization rate constant for slowly biodegradable material in an anoxic environment, K^1_{mpT}. It was also stated that the values of these constants depend upon the sewage source. At 20°C the values for μ_{nm20} ranged between 0,17 to

$0,65d^{-1}$ for f_{ca} between 0,05 and 0,24 and for K^{1}_{mp20} between 0,66 and 1,14 $mgCOD.mg\ X_a^{-1}d^{-1}$. Hence there is a need for an experimental method to determine these constants accurately and rapidly. Several experimental methods were investigated for this purpose :

(i) Series reactor systems under constant flow and load conditions.

(ii) Series reactor systems under cyclically varying flow and load conditions.

(iii) A single reactor system under constant flow and load conditions with alternating periods of an anoxic and of an aerobic environment in the reactor.

It was found that the first two systems, (i) and (ii), did not always give stable experimental result, due to small fluctuations in sewage characteristics or kinetic constants and to hydraulic effects.

Stable and reproducible values for the rates of nitrification and of denitrification were determined by using system (iii), i.e. the single reactor, alternatingly anoxic and aerobic system under constant flow and load conditions. Furthermore this system was found to be relatively easy to operate and to offer the possibility of accumulating a large number of relevant data during a short period of operation. This system appears to be the most convenient to determine nitrification and denitrification rates.

Using the single reactor system with alternating anoxic and aerobic periods, both nitrate concentration-time and alkalinity-time profiles were used to determine the rates of nitrification and denitrification. The relevant kinetic constant for nitrification, μ_{nmT}, was determined directly from the observed maximum nitrification rate. However, the relevant kinetic constant for denitrification, K^{1}_{mp}, could not be determined directly from the observed denitrification rate because this rate was due to utilization of both easily and slowly biodegradable material. In order to estimate the de-

nitrification rate due to utilization of easily biodegradable material in a single reactor system, the ratio $f_{ca} = S_{bsi}/S_{bi}$ had to be determined separately. This was achieved by operating an aerobic single reactor system under square wave flow and load conditions(12 h feed, 12 h no feed), with a short sludge age (< 3 days). Under these conditions, reduction of the oxygen uptake rate at the end of the feed period was shown to be wholly due to the termination of the easily biodegradable sewage source- a short sludge age was required to ensure that at feed termination the ammonia concentration was high and the nitrification rate did not decrease immediately after the end of the feed period. Usually at R_s < 3 days the nitrification rate and the associated oxygen uptake rate stay constant for some time after the end of the feed period.

With a known value of f_{ca} the value of the denitrification rate due to utilization of easily biodegradable material, dN_1/dt, was calculated as follows :

$$dN_1/dt = -((1 - PY_h)/2,86).f_{ca}.S_{bi}/R_h \qquad (6.18)$$

where R_h = hydraulic retention time in the single reactor. The denitrification rate due to utilization of slowly biodegradable material, (dN_2/dt),was then determined as the difference between the total rate of denitrification, r_{den}, and the denitrification rate due to utilization of easily biodegradable material :

$$dN_2/dt = r_{den} - dN_1/dt.$$

Using the general model, again $(dN_2/dt)$was expressed as :

$$dN_2/dt = -[K^1_{mpT}((1 - PY_h)/2,86)X_sP/(X_sP + K_{spT}X_a)]\ X_a \qquad (6.19)$$

whence the value of K^1_{mpT} was determined by trial and error computer simulations. It was found possible to calculate manually an approximated value for K^1_{mpT} by equating the Levenspiel ratio $X_sP/(X_sP + K_{spT}X_a)$ to unity.

This approximation is realistic when the value of X_s is high. A high value of X_s can be ensured by suitably proportioning the time fractions between the sequential anoxic and aerobic environments.

6. OPTIMIZATION OF NITROGEN REMOVAL

The general anoxic/aerobic activated sludge theory can predict the kinetic behaviour in single or series reactor processes under constant or variable flow and load conditions for any process already designed. However, the theory does not provide information to achieve optimal removal of nitrate in a particular situation and subject to the restrictions placed on the design. For this purpose an optimization procedure was developed which allows a rational selection of the relevant operational parameters of single sludge nitrification systems (i.e. sludge age; size and division of the anoxic sludge mass fraction; recycle ratios from the aerobic reactor and the settler to the predenitrification reactor) as a function of influent sewage characteristics and kinetic constants. Sewage characteristics and kinetic constants that were found to influence the optimal values of the operational parameters are:

(1) Influent flow and its variation over the day.

(2) Influent TKN concentration and its variation over the day.

(3) Influent COD concentration and its variation over the day.

(4) Composition of the influent COD (easily biodegradable fraction, f_{ca}, and unbiodegradable fractions, f_{us} and f_{up}.

(5) Temperature.

(6) Maximum specific growth rate of nitrifiers.

(7) Specific utilization rate constant for slowly biodegradable material in an anoxic environment.

The following considerations were fundamental to the development of the optimization procedure:

(1) Nitrification must be efficient.

(2) Denitrification must be maximum and, if possible, complete.

These two requirements are counteracting: Nitrification efficiency increases as the aerobic sludge mass fraction increases and the extent of denitrification increases as the anoxic sludge mass fraction increases. Because nitrification is a prerequisite for denitrification the first requirement - that of efficient nitrification - must always be satisfied. This, in a given design situation, sets a minimum aerobic and a consequential maximum anoxic sludge mass fraction. The maximum extent of denitrification in the anoxic reactor may or may not be sufficient to remove all the nitrate produced by nitrification. The extent of nitrification or nitrification capacity is linear with the influent TKN concentration, N_{ti}, whereas the maximum extent of denitrification or denitrification potential is proportional to the influent COD concentration, S_{ti}. Hence the relative magnitudes of the nitrification capacity and the denitrification potential in a particular design situation depend upon the influent TKN/COD ratio, (N_{ti}/S_{ti}). If the denitrification potential is greater than the nitrification capacity, complete denitrification is feasible. An expression was developed relating the maximum (N_{ti}/S_{ti}) ratio that allows complete denitrification, $(N_{ti}/S_{ti})_o$, to influent sewage characteristics, kinetic constants and operational parameters.

$$(N_{ti}/S_{ti})_o = \frac{(\alpha+K_2 C_r f_M)(1-f_{us}-P.f_{up})}{\frac{a}{a+s+1} + \frac{K_2}{K_3}\cdot\frac{(s+1)}{(a+s+1)}} + \frac{N_s + N_o + N_{ad}}{S_{ti}} \qquad (6.20)$$

where

f_M = maximum anoxic sludge mass fraction

a,s = recycle ratios from the aerobic reactor and the settler respectively

N_s = influent TKN concentration wasted with sludge (mg N.ℓ^{-1})

N_o = organic nitrogen effluent concentration (mg N.ℓ^{-1})

N_{ad} = ammonia effluent concentration (mg N.ℓ^{-1})

f_{us} = unbiodegradable soluble influent COD fraction (mg COD.mg COD^{-1})

f_{up} = unbiodegradable particulate influent COD fraction (mg VSS.mg COD^{-1})

Alternatively, Eq (6.20) can be used to determine (1) if complete denitrification for given sewage characteristics and kinetic constants

is possible and, if so, (2) the values of the operational parameters (sludge age and recycle ratios) for complete denitrification.

If complete denitrification is not possible, the objective is to maximize the extent of nitrate removal and hence minimize the effluent nitrate concentration, by suitably dividing the maximum anoxic sludge mass fraction over the pre- and post-denitrification reactors. For high (N_{ti}/S_{ti}) ratios the post-denitrification becomes counterproductive and for maximum nitrate removal the entire anoxic sludge mass fraction must be placed in the predenitrification reactor i.e. the optimal configuration is the Modified Ludzack Ettinger configuration. The minimum (N_{ti}/S_{ti}) ratio for which a Modified Ludzack Ettinger process is advantageous, is given by:

$$(N_{ti}/S_{ti})_1 = (\alpha+K_2C_rf_M)(1-f_{us}-P.f_{up})(a+s+1)/(a+s) + (N_s+N_o+N_{ad})/S_{ti}. \quad (6.21)$$

From Eqs (6.20 and 6.21), depending on the influent (N_{ti}/S_{ti}) concentration, three different situations may arise:

(1) Low (N_{ti}/S_{ti}) ratio, $(N_{ti}/S_{ti})<(N_{ti}/S_{ti})_o$: complete denitrification is possible, using a Bardenpho configuration.

(2) Medium (N_{ti}/S_{ti}) ratio : $(N_{ti}/S_{ti})_o<(N_{ti}/S_{ti})<(N_{ti}/S_{ti})_1$
Complete denitrification is not possible; maximum nitrate removal is achieved using a Bardenpho configuration. Denitrification is complete in the predenitrification reactor but not in the post-denitrification reactor.

(3) High (N_{ti}/S_{ti}) ratio: $(N_{ti}/S_{ti})>(N_{ti}/S_{ti})_1$
Complete denitrification is not possible; maximum nitrate removal is achieved using a Modified Ludzack Ettinger configuration.

From the work developed in this thesis it would appear that a design procedure is now available for single sludge nitrification denitrification systems that allows a rational selection of the operational parameters to affect maximum nitrogen removal for any waste flow with specified values for sewage characteristics and kinetic constants.

LIST OF REFERENCES

1. Alexander, W.V., Ekama, G.A. and Marais, G.v.R.(1979), "The Contact Stabilization Activated Sludge Process", Research Report No. W 28, Water Research Commission of South Africa, Pretoria, South Africa.

2. Andrews, J.F. and Busby, J.R. (1973), "Dynamic Modelling and Control Strategies for the Activated Sludge Process", Res.Rept., Dep. of Environ. Systems. Eng. Clemson Univ., Clemson, South Carolina, U.S.A.

3. Arkley, J.J. and Marais, G.v.R.(1981), "Effect of Anoxic Sludge Mass Fraction on the Activated Sludge Process", Res. Rep. W 38, Dept. Civil Eng. University of Cape Town.

4. Baas-Becking, L.G.M. and Parkes, S.G. (1927) "Energy Relations in the Metabolism of Autotrophic Bacteria", Physiol.Rev., 7, pp 85-106.

5. Balakrishnan and Eckenfelder (1970), "Nitrogen Removal by Modified Activated Sludge Process", J. Sanitary Engng Div. 96, 501-12.

6. Barnard, J.L. (1973) "Biological Denitrification", Water Pollut. Contr., 72, 6, pp 705-720.

7. Barnard, J.L. (1975),"Nutrient Removal in Biological Systems", Water Pollut. Contr., 74, pp 143-154.

8. Barnard, J.L. (1976),"A Review of Biological Phosphorus Removal in the Activated Sludge Process", Water S.A., 2, (3), 163.

9. Barth, Brenner and Lewis (1969),"Chemical-Biological Control of Nitrogen and Phosphorus in Waste-water Effluent", J.Wat.Pollut. Contr. Fed. 40, 2040-54.

10. Benninger R.W. and Sherrard, J.H., "Nitrification and Alkalinity Relationships in Activated Sludge", J.Wat.Pollut. Contr.Fed., 50, 2132 (1978).

11. Bishop, P.H. and Farmer, M., "The Fate of Nutrients during Aerobic Digestion", Jour.Environ.Eng.Div., Proc. Amer. Soc. Civil Engr., 104, 967, (1978).

12. Blackwell, L.G. (1971), "A Theoretical and Experimental Evaluation of the Transient Response of the Activated Sludge Process", PhD thesis, Clemson Univ., Clemson, S. Carolina,U.S.A.

13. Buswell, A.M., Shiota, T., Lawrence, N. and Van Meter, I. (1954), "Laboratory Studies on the Kinetics of Growth of Nitrosomonas with relation to the Nitrification Phase of the BOD Test", Applied Microbiology, 2, pp 21-25.

14. Banazzi, A. (1923), "On Nitrification, V. The Mechanism of Ammonia Oxidation", J.Bact., 8, p 343.

15. Boon, B. and Laudelout, H. (1962), "Kinetics of Nitrate Oxidation by *Nitrobacter winogradski*", Biochem. J., 85, pp 440-447.

16. Carlson, D.E. (1971), "Nitrogen Removal and Identification for Water Quality Control", OWRR Project No. 4040, Dept. Civil Eng., Univ. of Washington.

17. Chang, J.P. and Morris, J.E. (1962), "Studies on the Utilization of Nitrate by Micrococcus Denitrificans", J.Gen.Microbiol., 29, 301.

18. Christensen, M.H. and Harremoes, P. (1977), "Biological Denitrification of Sewage : A Literature Review", Prog.Wat.Tech., 8, 4/5, 509-555.

19. Christensen, M.H. and Marremoes, P. (1979), "Biological Denitrification in Water Treatment", Rept.72-2, Dept.Sanit.Eng., Tech. Univ. of Denmark.

20. Christensen, M.H. (1974), "Denitrification of Sewage by Alternating Process Operation". Presented: 7.IAWPR Conf., Paris (1974).

21. Delwiche, (1956), "Denitrification. A Symposium of Inorganic Nitrogen Metabolism. (Ed.: McElroy, Glass), Johns Hopkins Press (1956).

22. Dold, P.L., Ekama, G.A. and Marais, G.v.R. (1980), "A General Model for the Activated Sludge Process", Proc.10th Int.Conf.IAWPR, Toronto, in Prog.Wat.Tech., 12, 47-77.

23. Downing, A.L., Painter, H.A. and Knowles, G. (1964), "Nitrification in the Activated Sludge Process", J.Proc.Inst.Sew.Purif., 64,2, pp 130-158.

24. Downing, A.L., Tomlinson, T.G. and Truesdale, G.A. (1964), "Effects of Inhibitors on Nitrification in the Activated Sludge Process", J.Inst.Sew.Purif., 6, pp 537-550.

25. Downing, A.L. and Knowles, G., "Population Dynamics in Biological Treatment Plants". Presented at the 3rd Conference of the IAWPR, Munich, 1966.

26. Ekama, G.A. and Marais, G.v.R. (1978), "The Dynamic Behaviour of the Activated Sludge Process, Res.Rept.No. 27, Dept.Civil Eng., Univ. of Cape Town.

27. Ekama, G.A., van Haandel, A.C. and Marais, G.v.R. (1979), "The Present Status of Research on Nitrogen Removal : A Model for the Modified Activated Sludge Process". Presented at the Symposium on Nutrient Removal, C.S.I.R., Pretoria. To be published by the Water Research Commission of South Africa.

28. Ekama, G.A. and Marais, G.v.R. (1976), Unpublished research.

29. Engel, M.S. and Alexander, M., "Growth and Metabolism of *Nitrosomonas europeia*", Journal Bacteriology 76, 217-222 (1958).

30. Gasser, J.A., Chen, C.L. and Miele, R.P.,"Fixed Film Nitrification of Secondary Effluent", Journ.Environ.Eng.Div., Proc.Amer.Soc.Civil Eng., 104, EE 77 (1978).

31. Gould, G.W. and Lees, H. (1960), "Isolation and Culture of Nitrifying Organisms. Part I. *Nitrobacter*", Can.J.Micro., 6, p 299.

32. Gujer, W. (1977), "Design of Nitrifying Activated Sludge Process with the Aid of Dynamic Simulation", Prog.Wat.Tech., 9, pp 323-336,

33. Gujer, W. and Jenkins, D. (1975b), "A Nitrification Model for the Contact-Stabilization Activated Sludge Process", Water Research, 9, 5/6, pp 561-566.

34. Haug, R.T. and McCarty, P.L., "Nitrification with the Submerged Filter". Report prepared by the Department of Civil Engineering, Stanford University for the Environmental Protection Agency, Research Grant No. 17010 EPM, August, 1971.

35. Hamm, (1969), "Der Einfluss von Phosphatfallungsmitteln auf die Denitrifikation". (The influence on denitrification of phosphate precipitants). Z.Wasser - und Abwasser - Forschueng, 2 (1969), pp 180-2.

36. Heide, B.A. (1975), "Aerobe en Verdergaande Zuivering van Afvalwater in zeer Laagbelaste Actief-slib Systemen, Deel II", Rapport A 77, Inst. voor milieuhygiëne en gezondheidstechniek TNO, Delft. (in Dutch).

37. Heidman, J.A. (1979), "Sequential Nitrification-Denitrification in a Plug Flow Activated Sludge System", U.S. Environmental Protection Agency, Cincinnati, Ohio, Rept. EPA 600/2-79-157.

38. Hoffman, T. and Lees, H. (1953), "The Biochemistry of Nitrifying Organisms. 4. Respiration and Intermediary Metabolism of *Nitrosomonas*", Biochem.Jour., 54, pp.579-583.

39. Horstkotte, Niles, Parker and Caldwell, (1974), "Full-scale testing of a water reclamation system". J.WPC F 46 (1974) 181-197.

40. Huang, C.S. and Hopson, N.E. "Temperature and pH Effect on the Biological Nitrification Process". Presented at the Annual Winter Meeting, New York Water Pollution Control Association, New York City, January 1974.

41. Hunerberg and Sarfert, (1967). Versuche zur Stickstoffelimination aus dem Berliner Abwasser. (Experiments on the elimination of nitrogen from Berlin sewage). Gas Wasserfach Wasser-Abwasser, 108 (1967) pp 966-69 and 1197-1205.

42. Jacquart, J.C., Lefort, D. and Rovel, J.M. (1973), "An Attempt to Take Account of Biological Storage in the Mathematical Analysis of Activated Sludge Behaviour", Advances in Water Pollution Research (Ed. Jenkins, S.H.), Pergamon Press, New York.

43. Johnson, W.K. and Vania, P.J. (1971), "Nitrification and Denitrification of Wastewaters". San.Engng.Report No.1735, Inst. of Techn., Univ. of Minnesota.

44. Klapwijk,A. (1978), "Eliminatie van Stikstof uit Afvalwater door Denitrificatie", Doctoral Thesis Wageningen University, Wageningen, Netherlands. (In Dutch).

47. Knowles, G., Downing, A.L. and Barrett, M.J. (1965), "Determination of Kinetic Constants for Nitrifying Bacteria in Mixed Cultures with the Aid of an Electronic Computer", J.Gen.Micro., 38, pp 263-278.

48. Laudelout, H. and van Tichelen, L. (1960), "Kinetics of Nitrite Oxidation by *Nitrobacter winogradski*", J.Bacteriol., 79, pp. 39-42.

49. Lawrence, A.W. and McCarty, P.L. (1970), "Unified Basis for Biological Treatment Design and Operation", J.Sanit.Eng.Div., Proc.Am.Soc. Civil Eng., 96, SA3, pp. 757-778.

50. Lawrence, A.M. and Brown, C.G, "Biokinetic Approach to Optimal Design and Control of Nitrifying Activated Sludge Systems". Presented at the Annual Meeting of the New York Water Pollution Control Association, New York City, January 23, 1973.

51. Lees, H. (1954),"The Biochemistry of Nitrifying Bacteria", In: Symposium on Autotrophic Micro-organisms, pp. 84-98, Cambridge University Press, London.

52. Lees, H. and Simpson, J.R. (1957), "The Biochemistry of Nitrifying Organisms. 5. The Nitrite Oxidation by *Nitrobacter*", Biochem. J., 65, pp. 297-305.

53. Levenspiel, O. (1972), "Chemical Reaction Engineering", 2nd Ed., John Wiley, New York, pp. 465-469.

54. Lijklema, L. (1973), "Model for Nitrification in the Activated Sludge Process", Envir. Sci. and Tech., 7, 5, p.428.

55. Loehr, R.C., Prakasam, T.B.S., Srinath, E.G. and Joo, Y.D. (1973), "Development and Demonstration of Nutrient Removal from Animal Wastes". Report prepared for the Environmental Protection Agency, EPA-R2-73-095, January, 1973.

56. Loewenthal, R.E. and Marais, G.v.R. (1976), "Carbonate Chemistry of Aquatic Systems : Theory and Application", Ann Arbor Science, Michigan, U.S.A.

57. Loomis, T.A. (1974), "Essentials of Toxicology", 2nd Edition, Book published by Lea and Febiger, Philadelphia.

58. Loveless, J.E. and Painter, H.A. (1968), "The Influence of Metal Ion Concentration and pH Value in the Growth of a *Nitrosomonas* Strain Isolated from Activated Sludge", Jour.Gen.Micro., 52, pp.1-14.

59. Ludzack, F.J. and Ettinger, M.G. (1962), "Controlling Operation to Minimize Activated Sludge Effluent Nitrogen", J.Wat.Pollut.Control Fed., 34, 920-931.

60. Malan, W.N. and Gouws, E.P. (1966), "Geaktiveerde Slyk vir Rioolwatersuivering op Bellville", Research Report, Council for Scientific and Industrial Research, Nov., 1966. (In Afrikaans).

61. Marais, G.v.R. (1973) "The Activated Sludge Process at Long Sludge Ages", Res.Rept. No.W.3, Dept. of Civil Eng., Univ. of Cape Town.

62. Marais, G.v.R. and Ekama, G.A. (1976), "The Activated Sludge Process : Steady State Behaviour", Water S.A., 2, (4), 163-200.

63. Marsden, M.G. and Marais, G.v.R. (1976) "The role of the Primary Anoxic Reactor in Denitrification and Biological Phosphorus Removal", Res.Rept.No.19, Dept. Civil Eng., Univ. of Cape Town.

64. Matsche, (1972). The Elimination of Nitrogen in the Treatment Plant of Vienna-Blumenthal. Water Research, 6, (1972) pp 485-486.

65. McCarty, P.L. (1964), "Thermodynamics of Biological Synthesis and Growth", Proc. 2nd Inter.Confr. on Wat.Pollut. Res., 2, pp 169-199.

66. McCarty, P.L. (1972), "Energetics of Organic Matter Degradation", In: Water Pollution Microbiology, Edited by Mitchell, R., Wiley-Interscience.

67. McCarty, P.L. and Brodersen, C.F. (1962), "Theory of Extended Aeration Activated Sludge", J. Wat. Pollut. Control Fed., 34, 11, 1095-1103.

68. McKinney, R.E. (1962), "Mathematics of Complete Mixing Activated Sludge", J.San.Eng.Div. ASCE 88, SA3, Proc.Paper 3133, 87.

69. McKinney, R.E. and Symons, J.M. (1964), "Growth and Endogenous Metabolism: A Discussion, Proc. 1st Inter.Conf. on Water Pollution Control, London, 1962, Pergamon Press, Oxford, England.

70. McLaren, A.R. and Wood, R.J. (1976), "Effective Phosphorus Removal from Sewage by Biological Means", Water S.A., 2, 1, 47-50.

71. Melamed, A., Saliternik, C. and Wachs, A.M., "BOD Removal and Nitrification of Anaerobic Effluent by Activated Sludge". Presented at the 5th IAWPR Conference, San Francisco, California, July-August, 1970.

72. Meiklejohn, J. (1954), "Some Aspects of the Physiology of the Nitrifying Bacteria", In: Symposium on Autotrophic Micro-organisms, pp 68-83, Cambridge Univ. Press, London.

86. Poduska, R.A. and Andrews, J.F. (1974) "Dynamics of Nitrification in the Activated Sludge Process", Dept. of Environ. Systems Eng., Clemson Univ., Clemson, S. Carolina.

87. Pressly, T.A., Bishop, D.F., Pinto, A.P. and Cassel, A.F., "Ammonia Nitrogen Removal by Breakpoint Chlorination", Report prepared for the Environmental Protection Agency, Contract Number 14-12-818 February, 1973.

88. Sawyer, C.N., Wild, H.E. Jr., and McMahon, T.C., "Nitrification and Denitrification Facilities, Wastewater Treatment. Prepared for the EPA Technology Transfer Program, August, 1973.

89. Scearce, S.N., Benninger, B.W.,Weber, A.S. and Sherrard, Y.H. (1980), "Predictions of Alkalinity Changes in the Activated Sludge Process", J.Wat.Pollut.Control Fed., 52, pp 399-405.

90. Sehayek, E. and Marais, G.v.R. (1981), "Kinetics of Biological Nitrogen Removal in the Single Sludge Activated Sludge Process". Res.Rep.W37 Dept.Civil Eng., University of Cape Town.

91. Servizi, J.A. and Bogan, R.H. (1964), "Thermodynamic Aspects of Biological Oxidation and Synthesis", Jour.Wat.Pollut.Contr.Fed., 36, 5, pp 607-618.

92. Slechta, A.F. and Culp, G.L., "Water Reclamation Studies at the South Tahoe Public Utility District", J.Wat.Pollut.Control Fed.,39, 5, pp 787-814 (1967).

93. Schuster, (1970), "Laborversuche zur Elimination von Stickstoffverbindungen aus hauslichem Abwasser". (Laboratory experiments on removal of nitrogen compounds from domestic sewage). Fortschr.Wasserchem.Grenzg. 12 (1970) pp 139-148.

94. Skinner, F.A. and Walker, N. (1961), "Growth of *Nitrosomonas europea* in Batch and Continuous Cultures", Arch.Mikrobiol., 38, pp 339-349.

95. Stojanovic, B.J. and Alexander, M. (1958), "Effect of Inorganic Nitrogen on Nitrification", Soil Science, 6, pp 208-215.

96. Stenstrom, M.K. (1975), "A Dynamic Model and Computer Compatible Control Strategies for Wastewater Treatment Plants", PhD Thesis Clemson Univ., Clemson, S.Carolina, U.S.A.

97. Stenstrom, M.K. and Poduska, R.A. (1980), "The Effect of Dissolved Oxygen Concentration on Nitrification". Water Research, 14, 6, pp 645-650.

98. Stensel, H.D., Loehr, R.C. and Lawrence, A.W. (1973), "Biological Kinetics of Suspended Growth Denitrification", J.Wat.Pollut.Control Fed., 45, pp 249-261.

99. Stern, L.B. and Marais, G.v.R. (1974), "Sewage as Electron Donor in Biological Denitrification", Res.Rept.No.7, Dept. Civil Eng., Univ. of Cape Town.

100. Sutton, P.M., Murphy, K.L. and Jank, B.E., "Design Considerations for integrated Nutrient Removal Systems". Prog.Wat.Tech. 10 5/6, pp 469-478, Pergamon Press, London (1978).

101 Sutton, P.M., Jank, B.E., Monaghan, B.A. and Murphy, K.L. (1979), "Single Sludge Nitrogen Removal Systems", Research Report No.88, Environmental Protection Service, Environment CANADA.

102 Washington, D.R. and Hetling, L.J. (1965), "Volatile Sludge Accumulation in Activated Sludge Plants", J.Wat.Pollut.Control Fed.,37, 4, pp 499-507.

103 Wezernak, C.T. and Gammon, J.J. (1967), "Oxygen-Nitrogen Relationships in Autotrophic Nitrification". Appl.Microbiol., 15, pp 1211-1215.

104 U.S.EPA, "Process Design Manual for Nitrogen Control", Office of Technology Transfer, Washington DC, 1975.

105 Wilson, D.E. and Marais, G.v.R. (1976), "Adsorption Phase in Biological Denitrification", Res.Rept.No.11, Dept.Civil Eng., Univ. of Cape Town.

106 Wuhrmann, K. (1964), "Nitrogen Removal in Sewage Treatment Processes", Verh.Int.Ver.Theor.Angew.Limnol., 15, (in German).

APPENDICES

APPENDIX A1 : Listing of the computer program for dynamic steady state

APPENDIX A2 : Listing of the computer program for unsteady state

APPENDIX A3 : Listing of the results of the bench scale experiments

APPENDIX A4 : Listing of the results of the pilot scale experiments

APPENDIX A5 : Listing of the results of the alternating anoxic and aerobic reactor

APPENDIX A6 : Listing of the plotting programme

APPENDIX A1

COMPUTER PROGRAM FOR THE DYNAMIC STEADY STATE

```
C*************************************************************************
C
C        DYNAMIC STEADY STATE PROGRAM AS ON 05-05-1981
C
C*************************************************************************
C*************************************************************************
C*************************************************************************
C
C       ''THIS PROGRAMME IS THE JOINT PROPERTY OF THE
C       UNIVERSITY OF CAPE TOWN AND THE WATER RESEARCH
C       COMMISSION OF SOUTH AFRICA. BEFORE USER MAY
C       LEGALLY UTILIZE IT, HE MUST OBTAIN THE NECESSARY
C       AUTHORIZATION, I.E.:
C
C       (A) IN THE CASE OF STUDENTS AT THE UNIVERSITY OF
C           CAPE TOWN WHO WILL USE THE PROGRAMME FOR
C           EDUCATIONAL PURPOSES ONLY, THIS AUTHORIZATION
C           CAN BE OBTAINED FROM PROF. G. VAN R. MARAIS
C           OF THE UNIVERSITY OF CAPE TOWN, AND
C       (B) IN ALL OTHER CASES, WRITTEN AUTHORIZATION
C           MUST BE OBTAINED FROM BOTH THE UNIVERSITY
C           OF CAPE TOWN (PRIVATE BAG, RONDEBOSCH, 7700,
C           CAPE, SOUTH AFRICA) AND THE WATER RESEARCH
C           COMMISSION (P.O. BOX 824, PRETORIA, 0001,
C           SOUTH AFRICA)''
C
C*************************************************************************
C*************************************************************************
C
C       THIS PROGRAM IS USED TO SIMULATE THE DYNAMIC RESPONSE OF
C       SERIES REACTOR ACTIVATED SLUDGE SYSTEMS UNDER CYCLIC-
C       LY VARYING FLOW AND/OR LOAD CONDITIONS.THE REACTORS
C       ARE ASSUMED TO BE COMPLETELY MIXED . THE VALUES OF THE
C       VARIABLES ARE CALCULATED AS A FUNCTION OF TIME OVER
C       ONE CYCLE BY MEANS OF AN ITERATIVE CALCULATION PROCEDURE:
C       THE PROGRAM CALCULATES THE STARTING VALUES OF THE VARIABLES
C       FROM AN EQUIVALENT STEADY STATE PROCESS. THE STARTING VALUES
C       ARE USED IN AN ITERATIVE CALCULATION PROCESS TO CALCULATE THE
C       THE VALUES OF THE VARIABLES AS A FUNCTION OF TIME UNDER THE
C       PARTICULAR CYCLIC INPUT FLOW AND LOAD CONDITIONS. THE
C       EXPRESSIONS FOR CARBONACEOUS ENEGY REMOVAL AND REACTIONS
C       OF NITROGEN IN AN AEROBIC ENVIRONMENT ARE THOSE SUGGESTED
C       BY DOLD,EKAMA AND MARAIS. THE KINETIC EXPRESSIONS FOR
C       THESE REACTIONS IN AN ANOXIC ENVIRONMENT ARE IN TABLE 4.1
C       OF CHAPTER 4. ALKALINITY CHANGES ARE CALCULATED ON THE
C       BASIS THAT THIS PARAMENTER IS STOICHIOMETRICALLY AFFECTED
C       BY NITRIFICATION, DENITRIFICATION AND AMMONIFICATION.
C       THE CALCULATED VARIABLES ARE PRINTED AS A FUNCTION OF
C       TIME. THE VALUES OF THE MEASUREABLE VARIABLES CAN ALSO BE
C       WRITTEN TO A PLOT FILE FOR SUBSEQUENT PLOTTING TOGETHER
C       WITH EXPERIMENTALLY DETERMINED VALUES, USING THE PLOTTING
C       PROGRAM IN APPENDIX A6.
C       THE MAIN PROGRAM HAS SIX SUBROUTINES VIZ.
C            1.PAINCY : THIS SUBROUTINE CALCULATES THE SOLUTION
C                       OF A SINGLE REACTOR ACTIVATED SLUDGE SYSTEM
C                       UNDER EQUIVALENT CONSTANT FLOW AND LOAD
C                       CONDITIONS.
C            2.DSTRCY : CORRECTS FOR NONUNIFORM SLUDGE MASS DISTRI-
C                       BUTION IF THE UNDERFLOW IS NOT RECYCLED
C                       INTO THE FIRST REACTOR
C            3.DUMVCY : CALCULATES THE HYDAULIC EFFECTS OF INFLUENT
C                       AND RECYCLE FLOWS
C            4.AROBCY : CALCULATES THE DYNAMIC RESPONSE OF AN AERO-
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C                    BIC REACTOR.
C          5.ANOXCY : CALCULATES THE DYNAMIC RESPONSE OF AN ANOX-
C                     IC REACTOR.
C          6.SETLCY : CALCULATES THE DYNAMIC RESPONSE OF THE
C                     SETTLER.
C
C       THE PROGRAM AS LISTED BELOW IS SUBJECTED TO THE FOLLOWING
C       LIMITATIONS
C       1) A REACTOR IS EITHER CONTINOUSLY AEROBIC OR CONTINUOUSLY
C       ANOXIC
C       2)THE INFLUENT FLOW CAN ONLY BE INTRODUCED INTO ONE
C       REACTOR
C       3) THE RECYCLE FLOW FROM THE SETTLER CAN ONLY BE CONVEYED
C         TO ONE REACTOR (WHICH IS NOT NECESSARILY THE FIRST ONE)
C       4) IT IS POSSIBLE TO HAVE ONLY TWO INTERNAL RECYCLES FLOWS
C         (A AND B) EACH OF WHICH CONVEYS MIXED LIQUOR FROM ONE
C         REACTOR TO ANOTHER THE LATTER BEING NEARER TO THE REACTOR
C          RECIEVING THE INFLUENT
C       5) ALL RECYCLE FLOWS ARE CONSTANT WITH TIME.
C       6)OPTIONS FOR THE INFLUENT FLOW PATTERN ARE:
C          A)SINOIDAL
C          B)SQUARE WAVE
C          C)GENERAL PATTERN WITH LINEAR INTERPOLATION
C       7)THE CONCENTRATION PATTERNS OF INFLUENT COD,TKN,NITRATE
C         AND ALKALINITY HAVE THE SAME LIMITATIONS AS MENTIONED
C         FOR THE INFLUENT FLOW UNDER 6
C       8)IT IS ASSUMED THAT DURING A CYCLE THE CONSTANTS CHARA
C         CHARACTERIZING THE SEWAGE (FUS,FUP,FCA,FUN,FAN,SN) DO
C         NOT CHANGE
C       9)IT IS ASSUMED THAT THE D.O. CONCENTRATION IN THE AEROBIC
C         REACTORS IS SUFFICIENT SO THAT IT DOES NOT LIMIT THE
C         RATES OF UTILIZATION OF ORGANIC SUBSTRATE OR OF NITRI-
C         FICATION
C      10)IT IS ASSUMED THAT ANY D.O. RECYCLED TO THE ANOXIC
C         REACTORS DOES NOT AFFECT THE RATE OF DENITRIFICATION
C      11)THE PH AND TEMPERATURE ARE ASSUMED TO BE CONSTANT
C         OVER ONE CYCLE AND THE SAME IN ALL REACTORS
C      12)IS IS ASSUMED THAT THE PROCESS IS A SINGLE SLUDGE
C         PROCESS I.E. THERE IS ONLY ONE SETTLER
C      13) IT IS ASSUMED THAT SLUDGE WASTAGE TAKES PLACE FROM
C          ONE OR MORE REACTORS.THE FLOW OF WASTED SLUDGE IS CON-
C         STANT DURING THE PERIOD OF WASTAGE AND PROPORTIONAL
C         TO THE REACTOR VOLUME.
C       14) THE SETTLER IS TAKEN AS AN IDEAL LIQUID SOLID
C          SEPARATOR, I.E. THE RETENTION TIME IN THE SETTLER
C          IS ZERO AND NO REACTION TAKES PLACE IN THE SETTLER.
C
C          ALL THE LIMITATIONS ABOVE ,EXCEPT THE LAST ONE, ARE
C         IMPOSED TO REDUCE THE REQIRED CORE STORAGE. IF
C         REQUIRED THE PROGRAM CAN BE REWRITTEN IN SUCH A
C         FASHION THAT NONE OF THE LIMITATIONS 1 TO 13 APPLY.
C         THE FOURTEENTH LIMITATION CAN BE OMITTED ONLY IF
C         SUFFICIENT DATA IS AVAILABLE TO DESCRIBE THE
C         SETTLING BEHAVOUR UNDER CYCLIC FLOW AND LOAD
C         CONDITIONS.
C
C         NOTE THAT THIS PROGRAM CAN ONLY BE USED TO CHECK A
C         DESIGN AND HENCE REQUIRES A DESIGN CONFIGURATION
C         WITH SPECIFIED REACTOR VOLUMES.
C
C         THE PROGRAM HAS BEEN TESTED FOR DIFFERENT VARIANTS
C         OF THE ACTIVATED SLUDGE PROCESS, I.E.
C         1)SERIES SUSPENSION MIXED AERATED LAGOONS
C         2)CONTACT STABILIZATION PROCESS
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C           3)OXIDATION DITCH
C           4)SERIES COMPLETELY MIXED REACTOR SYSTEM(1 TO 6 REACTORS)
C           WITH AND WITHOUT ANOXIC REACTORS.
C
C       INPUT OF DATA
C       -------------
C
C       FIRST CARD:
C       -----------
C
C       TYPE = LOGICAL NUMBER DESCRIBING INFLUENT LOAD PATTERN
C         IF = 1    SINE WAVE LOAD PATTERN
C         IF = 2    SQUARE WAVE LOAD PATTERN
C         IF = 3    GENERAL LOADING PATTERN
C     RESULT = LOGICAL NUMBER TO DIRECT OUTPUT
C         IF = 1     PRINT RESULTS
C         IF = 2     PLOT RESULTS
C         IF = 0     PLOT AND PRINT RESULTS
C        KOM = TIME AT WHICH TIME BASE COUNTER COMMENCES
C         TN=TEST NUMBER
C       CRIT = MAXIMUM ERROR ALLOWED BETWEEN CONSECUTIVE DAYS'
C              VALUES OF PROCESS VARIABLES TO TEST FOR CONVERGENCE
C        NDH = LENGTH OF PERIOD FOR PLOTTING
C
C
C          SECOND CARD
C          -----------
C          (INPUT FLOW AND LOAD)
C         QO = AVERAGE INFLUENT FLOW FOR SINE WAVE PATTERN
C       STIO = INFLUENT COD CONCENTRATION DURING PEAK FLOW PERIOD FOR
C              SQUARE WAVE PATTERN OR AVERAGE FOR SINE WAVE PATTERN
C       NTIO = INFLUENT TKN CONCENTRATION DURING PEAK FLOW PERIOD FOR
C              SQUARE WAVE PATTERN OR AVERAGE FOR SINE WAVE PATTERN
C      NO3IO = INFLUENT NO3 CONCENTRATION DURING PEAK FLOW PERIOD FOR
C              SQUARE WAVE PATTERN OR AVERAGE FOR SINE WAVE PATTERN
C      ALKIO = INFLUENT ALK CONCENTRATION DURING PEAK FLOW PERIOD FOR
C              SQUARE WAVE PATTERN OR AVERAGE FOR SINE WAVE PATTERN
C     VARSTI = AMPLITUDE OF INPUT COD CONC. WAVE (SINE WAVE)
C     VARNTI = AMPLITUDE OF INPUT TKN CONC. WAVE (SINE WAVE)
C      VNO3I = AMPLITUDE OF INPUT NO3 CONC. WAVE (SINE WAVE)
C       VARQ = AMPLITUDE OF INFLUENT FLOW WAVE (SINE WAVE)
C      STI00 = INFLUENT COD CONCENTRATION DURING BASE FLOW PERIOD
C      NTI00 = INFLUENT TKN CONCENTRATION DURING BASE FLOW PERIOD
C     NO3I00 = INFLUENT NO3 CONCENTRATION DURING BASE FLOW PERIOD
C     ALKI00 = INFLUENT ALK CONCENTRATION DURING BASE FLOW PERIOD
C        LFP = LENGTH OF FEED PERIOD (SQUARE WAVE)
C         PQ = PEAK FLOW RATE IN SQUARE WAVE PATTERN
C         BQ = BASE FLOW RATE IN SQUARE WAVE PATTERN
C         VF = VOLUME OF FEED PER DAY
C      START = START OF FEEDING PERIOD( SQUARE WAVE)
C
C       THIRD CARD
C       ----------
C       (KINETIC CONSTANTS AND MASS PARAMETERS)
C
C        AN EXTRA T OR 20 INDICATES VALUES AT T OR 20 DEG. C
C         KV = MAX. RATE OF COD TRANSFER FROM SEWAGE INTO STORAGE
C         KR = RATE OF CONVERSION OF ORGANIC N TO SALINE NH3
C        KMP = MAX. RATE OF  ORGANISM SYNTHESIS FROM STORED COD
C        KMS = MAX. RATE OF  ORGANISM SYNTHESIS FROM SOLUBLE COD
C        KSP = HALF SATURATION COEF. FOR STORED COD  UTILIZATION
C        KSS = HALF SATURATION COEF. FOR SOLUBLE COD  UTILIZATION
C         UM = MAXIMUM GROWTH RATE OF NITRIFYING ORGANISMS
C         KN = HALF SATURATION COEF. FOR NH3  UTILIZATION
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C       BH = DEATH RATE
C       BN = DEATH RATE FOR NITRIFICATION
C       YH = YIELD OF ORGANISMS FROM COD UTILIZED
C       YN = ACTIVE MASS YIELD OF NITRIFYING ORGANISMS FROM AMMONIA
C        F = UNBIODEG. FRACTION OF ORGANISM
C       FR = MAXIMUM FRACTION OF STORED COD (AS VSS) TO ACTIVE
C            ORGANISM MASS
C       FN = FRACTION OF CARBONACEOUS CELL MASS AS N
C      FNS = FRACTION OF NITROGEN ASSOCIATED WITH STORED COD
C        P = COD EQUIVALENT OF VSS
C      FCA = RATIO OF EASILY (SOLUBLE) TO TOTAL INF. BIODEG. MAT.
C      FOE = FRACTION OF N RELEASED BY DEATH
C            AS ORGANIC NITROGEN
C      FOS = FRACTION OF NITROGEN REQUIRED FOR CELL SYNTHESIS
C            AS ORGANIC NITROGEN
C     FACT = RATIO OF KMP IN AN ANOXIC AND IN AN AEROBIC ENV.
C      FCS = FRACTION OF LYSED BIODEG. MATERIAL RELEASED AS
C            EASILE BIODEGRADABLE MATERIAL
C
C
C       FOURTH CARD
C       -----------
C       (OPERATIONAL PARAMETERS AND SEWAGE CHARACTERISTICS)
C          RS = SLUDGE AGE
C           NR= NUBER OF REACTORS IN SERIES
C           D  = NUMER OF INTEGRATION STEPS PER DAY
C           NJ = NUMBER OF SUBSTEP FOR UTILIZARION OF SBS AND NH3
C         FUS = FRACTION OF SOLUBLE UNBIODEG. COD
C         FUP = FRACTION OF SOLID UNBIODEG. COD AS VSS
C          UN = UNBIODEGRADABLE FRACTION OF N
C        SN = FRACTION OF TKN AS AMMONIA
C
C       FIFTH CARD
C       ----------
C       (TEMPERATURE DEPENDENCIES , PH AND TEMP.)
C
C       THEA = ARRHENIUS TEMPERATURE DEP. COEF. FOR KMS
C       THEM = ARRHENIUS TEMPERATURE DEP. COEF. FOR KMP
C       THES = ARRHENIUS TEMPERATURE DEP. COEF. FOR KSP
C       THEE = ARRHENIUS TEMPERATURE DEP. COEF. FOR BHT
C       THEN = ARRHENIUS TEMPERATURE DEP. COEF. FOR UNMT
C       THEZ = ARRHENIUS TEMPERATURE DEP. COEF. FOR KSS
C       PHIN = PH DEPENDENCY OF NITRIFICATION RATE
C       TEMP = MIXED LIQUOR TEMPERATURE
C         PH = MIXED LIQUOR PH
C
C       SIXTH CARD
C       ----------
C       (REACTOR VOLUMES AND SLUDG WASTAGE)
C
C        V = VOLUME OF REACTOR
C     LRSD = LOGICAL NUMBER DESCRIBING SLUDGE WASTAGE
C            FROM REACTORS
C       IF = 1  SLUDGE TO BE WASTED FROM REACTOR
C       IF = 0  NO SLUDGE TO BE WASTED FROM REACTOR
C
C     NTDN = LOGICAL NUMBER TO INDICATE ENVIRONMENT IN A REACTOR
C       IF = 1 REACTOR IS AEROBIC
C       IF = 0 REACTOR IS ANOXIC
C
C
C       SEVENTH CARD
C       ------------
C
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256       C       ICSD = TIME AT WHICH SLUDGE WASTAGE COMMENCES
257       C       ISSD = TIME AT WHICH SLUDGE WASTAGE TERMINATES
258       C       NRSD = NUMBER OF REACTORS FROM WHICH SLUDGE IS WASTED
259       C       LINI = REACTOR NUMBER RECIEVING INFLUENT FLOW
260       C       LSRI = REACTOR NUMBER RECIEVING UNDERFLOW
261       C       LARI = REACTOR NUMBER RECIEVING A-RECYCLE
262       C       LBRI = REACTOR NUMBER RECIEVING B-RECYCLE
263       C       LARO = REACTOR NUMBER FROM WHICH LEAVES A-RECYCLE
264       C       LBRO = REACTOR NUMBER FROM WHICH LEAVES B-RECYCLE
265       C         SR = UNDERFLOW RECYCLE  RATIO
266       C         AR = A-RECYCLE RATIO
267       C         BR = B-RECYCLE RATIO
268       C
269       C       DIMENSION STATEMENTS
270       C       --------------------
271       C
272             PARAMETER NE=6
273             PARAMETER NEE=7
274             PARAMETER ND=241
275             PARAMETER NDD=242
276             COMMON/EXTC/SBS(NE,ND),SU(NE,ND),XA(NE,ND),XC(NE,ND),XI(NE,ND),
277            1XV(NE,ND),XS(NE,ND),NU(NE,ND),NON(NE,ND),XN(NE,ND),NH3(NE,ND),
278            2NO3(NE,ND),SBP(NE,ND),ALK(NE,ND)
279             COMMON/BLOCKC/Q(NEE,NDD),V(NE),SBI(ND),SUI(ND),NIT(NE,ND),
280            1NUI(ND),XII(ND),NONI(ND),NH3I(ND),NO3I(ND),SBPR(ND),ALKI(ND),
281            2XAR(ND),XSR(ND),XER(ND),XIR(ND),XNR(ND),SYNTH(NE,ND),QAV(NEE),
282            3ANPOT(NE,ND),AMON(NE,ND),DSFCTR(NE)
283             CHARACTER IMFT*19/'((20X,14, (E13.5)))'/,CH*1
284             REAL KVT,KMPT,KSPT,KMST,KSST,KRT,KNT
285             REAL NUI,NONI,NH3I,NO3I,NIT
286             REAL NU,NON,NH3,NO3
287             INTEGER D,DD,NK,TYPE,RESULT,TN
288             DOUBLE PRECISION DT,DDT,RDT
289             DIMENSION STI(ND),STQ(ND)
290             REAL NTI(ND),NTQ(ND),NO3Q(ND),NXII(ND)
291             REAL NTIAV,NUIAV,NONIAV,NO3IAV,NH3IAV,NXIIAV
292             REAL NTQAV,NO3QAV
293             REAL NTIMAX,NO3IAX,NTQMAX,NO3QAX
294             REAL NTI0,NTI00,NO3I0,NO3I00,LFP
295             DIMENSION RA(NE),RN(NE),LRSD(NE),NTDN(NE),ALKQ(ND)
296             REAL KV20,KMP20,KSP20,KMS20,KSS20,KNMAX,KN20,KR20
297             DIMENSION STOSBP(NE,ND),STOXV(NE,ND),STNO3(NE,ND),STOXN(NE,ND)
298             DIMENSION KO(ND),WQ(NE,ND),STOXS(NE,ND),
299            1OC(NE,ND),ON(NE,ND),OT(NE,ND),ST(NE,ND),RXS(NE,ND)
300             REAL NT(NE,ND)
301             DIMENSION SBSAV(NE),ODAV(NE),OCAV(NE),RNO3(NE),ALKAV(NE),AMAV(NE),
302            1ONAV(NE),OTAV(NE),STAV(NE),SUAV(NE),SBPAV(NE),
303            2XAAV(NE),XEAV(NE),XSAV(NE),XIAV(NE),XVAV(NE),
304            3XNAV(NE),RXSAV(NE),NAME(NE),LNAME(2),APTAV(NE),APTNR(NE)
305             REAL NONAV(NE),NTAV(NE),NO3AV(NE),NH3AV(NE),NUAV(NE),NSWAV(NE)
306             REAL NONSS,NUSS,NO3SS,NH3SS
307       C
308       C             READ IN INPUT DATA
309       C             ------------------
310       C
311         100 FORMAT ( )
312         101 FORMAT(2I1,I4,A2,I1,F5.3)
313             READ 101,TYPE,RESULT,KOM,TN,NDH,CRIT
314             IF(TYPE.EQ.1) GO TO 111
315             IF(TYPE.EQ.2) GO TO 112
316             IF(TYPE.EQ.3) GO TO 113
317         111 READ 100,PI
318             READ 100,Q0,STI0,NTI0,NO3I0,ALKI0,VARQ,VARSTI,VARNTI,VNO3I,VALKI
319             GO TO 113
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      112 READ 100,VF,STIO,NTIO,NO3IO,ALKIO,RQ,STI00,NTI00,NO3I00,ALKI00,
     &LFP,START
      113 READ(8,100) KV20,KMP20,KSP20,KMS20,KSS20,UMAX,KNMAX,BH20,BN20,
     1YH,YN,F,P,FN,FOE,FACT,FCS,KR20,FOS,FCA,FR
          READ 100,RS,NR,D,NJ,FUS,FUP,UN,SN
          READ 100,THEA,THEM,THES,THEE,PHIN,THEN,THEZ,TEMP,PH
          READ(8,100) (V(J),LRSD(J),NTDN(J),J=1,NR,1)
          READ(8,100) ICSD,ISSD,NRSD,LIRI,LSRI,LARI,LARO,LBRI,LBRO,SR,AR,BR
          READ(8,100) (Q(1,I),I=1,NL,KH)
          READ(8,100) (STI(I),I=1,NL,KH)
          READ(8,100) (NTI(I),I=1,NL,KH)
          READ(8,100) (NO3I(I),I=1,NL,KH)
          READ(8,100) (ALKI(I),I=1,NL,KH)
C
C         CALCULATION CHARACTERISTICS
C         ---------------------------
C         THIS PROGRAMME REQUIRES THE FOLLOWING CORE STORAGE CAPACITY:-
C          FOR ND=241 AND NF=4*(ND-1)+1 AND NEE=NE+1
C          STORAGE=10*NE+23 KILOWORDS
C          DOUBLING ND INCREASES STORAGE REQUIREMENT BY 1.75
C          APPROX. CALCULATION CPU TIME ON A UNIVAC 1106 COMPUTER
C          FOR 100 ITERATIONS:
C          CPU TIME (SECONDS) =50*NE+30
C
C          TN = NUMBER OF TEST CALCULATION
C          KO = TIME BASE VARIABLE
C           M = CYCLE NUMBER MONITOR
C           D = NUMBER OF INTERGRATION STEPS PER DAY
C          DT = INTERGRATION STEP LENGTH
C          DD = NUMBER OF INTERGRATION STEPS PER DAY FOR NITRIFICATION
C         DDT = INTERGRATION STEP LENGTH FOR NITRIFICATION
C         RDT = RATIO OF INTERGRATION STEP LENGTHS
C          KH = NUMBER OF TIME STEPS PER HOUR
C          HK = NO. OF INTERVALS PER DAY  OF SLUDGE WASTAGE
C         DDT = INTERGRATION STEP LENGTH FOR NITRIFICATION
C         RDT = RATIO OF INTERGRATION STEP LENGTHS
C          KH = NUMBER OF TIME STEPS PER HOUR
C          HK = NO. OF INTERVALS PER DAY  OF SLUDGE WASTAGE
C
C         SEWAGE CHARACTERISTICS
C         ----------------------
C           Q = INFLUENT FLOW
C         STI = TOTAL INFLUENT COD
C         NTI = TOTAL INFLUENT TKN
C         STQ = TOTAL INFLUENT COD LOAD
C         NTQ = TOTAL INFLUENT TKN LOAD
C        NO3Q = TOTAL INFLUENT NO3 LOAD
C        NONI = INFLUENT ORGANIC NITROGEN CONCENTRATION
C        NH3I = INFLUENT AMMONIA CONCENTRATION
C        NO3I = INFLUENT NITRATE CONCENTRATION
C         NUI = UNBIODEGRADABLE FRACTION OF SEWAGE N
C        NXII = UNSABLE N IN INERT MATERIAL IN INFLUENT
C         XII = UNBIODEG. SOLID FRACTION OF SEWAGE COD AS VSS
C         SBI = BIODEG. FRACTION OF SEWAGE COD
C         SUI = UNBIODEG. SOLUBLE FRACTION OF SEWAGE COD
C       NM,NN = LIMITS OF PEAK FLOW PERIOD
C        QMAX = MAXIMUM FLOW VALUE OF THE DAY
C      STIMAX = MAXIMUM COD CONCENTRATION VALUE OF THE DAY
C      NTIMAX = MAXIMUM TKN CONCENTRATION VALUE OF THE DAY
C      NO3IAX = MAXIMUM NO3 CONCENTRATION VALUE OF THE DAY
C      STQMAX = MAXIMUM COD LOAD VALUE OF THE DAY
C      NTQMAX = MAXIMUM TKN LOAD VALUE OF THE DAY
C      NO3QAX = MAXIMUM NO3 LOAD VALUE OF THE DAY
C      VARSTL = AMPLITUDE OF INFLUENT COD LOAD WAVE (SINE WAVE)
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384       C      VARNTL = AMPLITUDE OF INFLUENT TKN LOAD WAVE (SINE WAVE)
385       C      VRNO3L = AMPLITUDE OF INFLUENT NO3 LOAD WAVE (SINE WAVE)
386       C
387       C         PROCESS CHARACTERISTICS
388       C         -----------------------
389       C
390       C        TEMP = OPERATING TEMPERATURE OF MIXED LIQUOR
391       C          PH = OPERATING PH OF MIXED LIQUOR
392       C         RSM = MINIMUM SLUDGE AGE FOR NITRIFICATION
393       C          RN = NOMINAL HYDRAULIC RETENTION TIME IN REACTORS
394       C          RA = ACTUAL HYDRAULIC RETENTION TIME IN REACTORS
395       C         TRN = TOTAL HYDRAULIC RETENTION TIME OF PROCESS
396       C         QSR = SLUDGE UNDERFLOW RECYCLE RATE
397       C         QAR = MIXED LIQUOR RECYCLE RATE (A-RECYCLE)
398       C         QBR = MIXED LIQUOR RECYCLE RATE (B-RECYCLE)
399       C          NR = NUMBER OF REACTORS IN SERIES
400       C         NRR = NUMBER OF REACTORS IN SERIES INCLUDING CLARIFIER
401       C          VP = TOTAL VOLUME OF PROCESS
402       C          WQ = SLUDGE WASTE FLOW
403       C
404       C      NOTE
405       C      ----
406       C         NITROGEN CONCENTRATIONS MEASURED IN MG-N/L WITH TKN TEST
407       C         COD CONCENTRATIONS MEASURED IN MG/L OF COD
408       C         VSS CONCENTRATIONS MEASURED IN MG/L OF VSS
409       C         OXYGEN CONSUMPTIONS IN MG/L/HOUR
410       C         UNIT OF MASS IS THE MG
411       C         UNIT OF VOLUME IS THE L
412       C         UNIT OF TIME IS THE DAY
413       C         SYMBOL S REFERS TO COD
414       C         SYMBOL X REFERS TO VSS
415       C         SYMBOL N REFERS TO NITROGEN
416       C         SYMBOL O REFERS TO OXYGEN
417       C
418       C
419       C         GENERAL NOTATION FOR PROCESS VARIABLES IN REACTOR
420       C         ------------------------------------------------
421       C
422       C         SBP = BIODEGRADABLE PARTICULATE COD
423       C         SBS = BIODEGRADABLE SOLUBLE COD
424       C          SU = UNBIODEGRADABLE COD
425       C          ST = TOTAL COD
426       C          XS = STORED COD
427       C          XA = ACTIVE ORGANISMS
428       C          XE = INERT RESIDUE FROM DECAYING ORGANISMS
429       C          XI = INERT MATERIAL FROM SEWAGE
430       C          XN = ACTIVE NITROSOMONAS
431       C          XV = TOTAL VSS
432       C          OA = OXYGEN CONSUMPTION RATE FOR ADSORPTION OF COD
433       C          OS = OXYGEN CONSUMPTION RATE FOR SYNTHESIS
434       C          OE = OXYGEN CONSUMPTION RATE FOR ENDOGENOUS RESPIRATION
435       C          OC = TOTAL CARBONACEOUS OXYGEN CONSUMPTION RATE
436       C          ON = OXYGEN CONSUMPTION FOR NITRIFICATION
437       C          OT = TOTAL OXYGEN CONSUMPTION RATE
438       C        AMON = AMMONIFICATION RATE (MG N/L/D)
439       C         NSW = NITROGEN REMOVED IN SLUDGE WASTAGE PER DAY
440       C          NT = TOTAL TKN IN REACTOR
441       C          NU = UNBIODEGRADABLE TKN IN REACTOR
442       C         NON = ORGANIC NITROGEN CONCENTRATION  IN REACTOR
443       C         NH3 = AMMONIA CONCENTRATION IN REACTOR
444       C         NO3 = NITRATES CONCENTRATION IN REACTOR
445       C         ALK = ALKALINITY CONCENTRATION IN REACTOR
446       C         SCI = ENERGY ENTERING STORAGE FROM LIQUID PHASE (MG-COD/L)
447       C        SCVO = ENERGY REMOVED FROM STORAGE DUE TO CELL SYNTHESIS
```

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C
C        THE SUFFIX 'AV' AFTER A SYMBOL INDICATES DAILY AVERAGE
C         VALUES FOR VARIABLES IN INFLUENT OR REACTORS
C        THE SUFFIX 'R' AFTER A SYMBOL INDICATES
C         VALUES FOR VARIABLES IN RECYCLE
C        THE SUFFIX 'P' AFTER A SYMBOL INDICATES VALUES
C         FOR VARIABLES CARRIED OVER TO PLOTTER ROUTINES
C        THE SUFFIX 'MAV' AFTER A SYMBOL INDICATES DAILY AVERAGE
C         VALUES OF VARIABLES FOR PROCESS
C
C       BALC = PERCENTAGE COD RECOVERY
C      TCODI = TOTAL COD MASS INPUT
C      TCODO = TOTAL COD MASS OUTPUT
C      TNTI = TOTAL TKN MASS INPUT
C      TNTO = TOTAL TKN MASS OUTPUT
C      TALKI = TOTAL ALKALINITY MASS INPUT
C      TALKO = TOTAL ALKALINITY MASS OUTPUT
C      TWCOD = TOTAL COD MASS IN WASTE FLOWS FROM REACTORS
C      TWN  = TOTAL TKN MASS IN WASTE FLOWS FROM REACTORS
C      TWALK = TOTAL ALK MASS IN WASTE FLOWS FROM REACTORS
C       WCOD = COD MASS IN WASTE FLOW FROM REACTOR
C       ECOD = TOTAL COD MASS IN EFFLUENT
C       BALN = PERCENTAGE NITROGEN RECOVERY
C       TNTI = TOTAL NITROGEN MASS INPUT
C       TNTO = TOTAL NITROGEN MASS OUTPUT
C        TWN = TOTAL NITROGEN MASS IN WASTE FLOWS FROM REACTORS
C         WN = NITROGEN MASS IN WASTE FLOW FROM REACTOR
C       ECOD = TOTAL COD IN EFFLUENT
C         EN = TOTAL NITROGEN MASS IN EFFLUENT
C       EALK = TOTAL ALKALINTY IN EFFLUENT
      KH=D/24
      NL=D+1
      NLL=D+2
      DD=NJ*D
      NRR=NR+1
      IF(TYPE.EQ.1) GO TO 115
      IF(TYPE.EQ.2) GO TO 116
      IF(TYPE.EQ.3) GO TO 117
  115 DO 8 I=1,NL,KH
      KL=(I-1)/KH
      DUM=SIN(2*PI*(KL-8)/24)
      Q(1,I)=Q0*(1+VARQ*DUM)
      STI(I)=STI0*(1+VARSTI*DUM)
      NTI(I)=NTI0*(1+VARNTI*DUM)
      NO3I(I)=NO3I0*(1+VNO3I*DUM)
      ALKI(I)=ALKI0*(1+VALKI*DUM)
    8 CONTINUE
C
C        INTERPOLATION OF STI, NO3I, Q AND NTI AT REQUIRED INTERVALS
C        ------------------------------------------------------------
C
  117 DO 25 I=1,24
      KI=KH*I+1
      KJ=KH*(I-1)+1
      DIFSTI=(STI(KI)-STI(KJ))/KH
      DIFFQ=(Q(1,KI)-Q(1,KJ))/KH
      DIFNTI=(NTI(KI)-NTI(KJ))/KH
      DNO3I=(NO3I(KI)-NO3I(KJ))/KH
      DALKI=(ALKI(KI)-ALKI(KJ))/KH
      DO 25 J=2,KH,1
      KK=KJ+J-1
      STI(KK)=STI(KJ)+(J-1)*DIFSTI
      Q(1,KK)=Q(1,KJ)+(J-1)*DIFFQ
      NTI(KK)=NTI(KJ)+(J-1)*DIFNTI
```

```
512             NO3I(KK)=NO3I(KJ)+(J-1)*DNO3I
513             ALKI(KK)=ALKI(KJ)+(J-1)*DALKI
514          25 CONTINUE
515             GO TO 118
516         116 PQ=BQ+(VF-BQ)*24.0/LFP
517             DO 24 I=1,NL,1
518             NTI(I)=NTI00
519             STI(I)=STI00
520             Q(1,I)=BQ
521             NO3I(I)=NO3I00
522             ALKI(I)=ALKI00
523          24 CONTINUE
524             NM=START*KH+1
525             END=START+LFP
526             NN=LFP*KH+NM
527             DO 23 I=NM,NN,1
528             STI(I)=STI0
529             Q(1,I)=PQ
530             NTI(I)=NTI0
531             NO3I(I)=NO3I0
532             ALKI(I)=ALKI0
533          23 CONTINUE
534         118 DO 1 I=1,NL
535             STQ(I)=STI(I)*Q(1,I)
536             NTQ(I)=NTI(I)*Q(1,I)
537             NO3Q(I)=NO3I(I)*Q(1,I)
538             ALKQ(I)=ALKI(I)*Q(1,I)
539             SUI(I)=FUS*STI(I)
540             NUI(I)=UN*NTI(I)
541             XII(I)=FUP*STI(I)
542             NXII(I)=XII(I)*FN
543             NH3I(I)=SN*NTI(I)
544             NONI(I)=NTI(I)-NUI(I)-NXII(I)-NH3I(I)
545           1 SBI(I)=STI(I)*(1.0-FUS-FUP*P)
546          C
547          C       CALCULATION OF AVERAGE SEWAGE CHARACTERISTICS
548          C       ---------------------------------------------
549          C
550             SUMQ=0
551             SUMSTI=0
552             SUMNTI=0
553             SNO3I=0.0
554             SALKI=0.0
555             QMAX=0.0
556             STIMAX=0.0
557             SUMSTQ=0.0
558             STQMAX=0.0
559             SUMNTQ=0.0
560             NTQMAX=0.0
561             NTIMAX=0.0
562             SNO3Q=0.0
563             NO3QAX=0.0
564             NO3IAX=0.0
565             SALKQ=0.0
566             ALKQAX=0.0
567             ALKIAX=0.0
568             DO 2 I=1,D,1
569             SUMNTI=SUMNTI+NTI(I)
570             SNO3I=SNO3I+NO3I(I)
571             SALKI=SALKI+ALKI(I)
572             SUMSTI=SUMSTI+STI(I)
573             SUMSTQ=SUMSTQ+STQ(I)
574             SUMNTQ=SUMNTQ+NTQ(I)
575             SNO3Q=SNO3Q+NO3Q(I)
```

```
576             SALKQ=SALKQ+ALKQ(1)
577             IF(STQ(1).GT.STQMAX) STQMAX=STQ(I)
578             IF(NTQ(1).GT.NTQMAX) NTQMAX=NTQ(I)
579             IF(NO3Q(I).GT.NO3QAX) NO3QAX=NO3Q(I)
580             IF(ALKQ(I).GT.ALKQAX) ALKQAX=ALKQ(I)
581             IF(Q(1,I).GT.QMAX) QMAX=Q(1,I)
582             IF(STI(I).GT.STIMAX) STIMAX=STI(I)
583             IF(NTI(I).GT.NTIMAX) NTIMAX=NTI(I)
584             IF(NO3I(I).GT.NO3IAX) NO3IAX=NO3I(I)
585             IF(ALKI(I).GT.ALKIAX) ALKIAX=ALKI(I)
586           2 SUMQ=SUMQ+Q(1,I)
587             QAV(1)=SUMQ/D
588             STQAV=SUMSTQ/D
589             NTQAV=SUMNTQ/D
590             NO3QAV=SNO3Q/D
591             ALKQAV=SALKQ/D
592             IF(TYPE.EQ.2) GO TO 119
593             NTIAV=SUMNTI/D
594             STIAV=SUMSTI/D
595             NO3IAV=SNO3I/D
596             ALKIAV=SALKI/D
597             GO TO 120
598         119 STIAV=STQAV/VF
599             NTIAV=NTQAV/VF
600             NO3IAV=NO3QAV/VF
601             ALKIAV=ALKQAV/VF
602         120 SBIAV=STIAV*(1.0-FUS-FUP*P)
603             SUIAV=FUS*STIAV
604             NUIAV=NTIAV*UN
605             XIIAV=FUP*STIAV
606             NXIIAV=FN*XIIAV
607             NH3IAV=SN*NTIAV
608             NONIAV=NTIAV-NXIIAV-NUIAV-NH3IAV
609             QSR=SR*QAV(1)
610             QAR=AR*QAV(1)
611             QBR=BR*QAV(1)
612             IF(TYPE.EQ.2) GO TO 121
613             VARSTL=STQMAX/STQAV-1.00
614             VARNTL=NTQMAX/NTQAV-1.00
615             VRNO3L=NO3QAX/NO3QAV-1.00
616             VRALKL=ALKQAX/ALKQAV-1.00
617             IF(VRNO3L.LT.0.0) VRNO3L=0.0
618             IF(VRALKL.LT.0.0) VRALKL=0.0
619             IF(TYPE.EQ.1) GO TO 121
620             VARQ=QMAX/QAV(1)-1.00
621             VARSTI=STIMAX/STIAV-1.00
622             VARNTI=NTIMAX/NTIAV-1.00.
623             VNO3I=NO3IAX/NO3IAV-1.00
624             VALKI=ALKIAX/ALKIAV-1.00
625             IF(VNO3I.LT.0.0) VNO3I=0.0
626             IF(VALKI.LT.0.0) VALKI=0.0
627         121 CONTINUE
628       C
629       C              CALCULATE TOTAL AND AEROBIC VOLUMES
630       C              -----------------------------------
631       C
632             VP=0.0
633             VPA=0.00
634             DO 13 J=1,NR,1
635             VP=VP+V(J)
636             IF(NTDN(J).EQ.2) GO TO 13
637             VPA=VPA+V(J)
638          13 CONTINUE
639       C
```

```
      C             ADJUST TEMPERATURE AND PH DEPENDENT CONSTANTS
      C             ---------------------------------------------
      C
            IF(PH.LT.8.0) UM20=UMAX
            IF(PH.LT.7.2) UM20=UMAX*(PHIN)**(PH-7.20)
            IF(PH.LT.8.0) KN20=KNMAX
            IF(PH.LT.7.2) KN20=KNMAX*(PHIN)**(7.20-PH)
            UMT=UM20*(THEN)**(TEMP-20.0)
            KNT=KN20*(THEN)**(TEMP-20.0)
            KSPT=KSP20*(THES)**(20.0-TEMP)
            KMPT=KMP20*(THEM)**(TEMP-20.0)
            KSST=KSS20*(THEZ)**(TEMP-20.0)
            KMST=KMS20*(THEA)**(TEMP-20.0)
            IF(NTDN(1).EQ.1) KMST=10.0
            KRT=KR20*(THER)**(TEMP-20.0)
            KVT=KV20*(THER)**(TEMP-20.0)
            BHT=BH20*(THEB)**(TEMP-20.0)
            BNT=BN20*(THEB)**(TEMP-20.0)
            RSM=1.0/((UMT/(1.0+(KNT/(SM*NTIAV))))-BNT)
            RSA=VPA/VP*RS
            TRN=VP/QAV(1)
            FNS=NONIAV/SBIAV
            CC=(1.0-F)*BHT*YH*RS/(1.0+BHT*(1.0-YH*F*(1.0-F))*RS)
            FNS=(FNS*(1.0-FCA)+FN*CC)/((1.0-FCA)+F*CC)
      C
      C                 CALCULATE INITIAL CONDITIONS (PARTICULAR INTEGRAL)
      C*****************************************************************
            CALL PAINCY(YH,FCA,RS,RSA,BHT,KVT,TEN,KMPT,KSPT,KMST,KSST,
           1P,F,FN,FNS,KRT,KNT,BNT,UMT,YN,NITDEN,SR,AR,BR,FACT,FR,
           2SBIAV,SUIAV,XIIAV,NUIAV,NONIAV,NTIAV,NO3IAV,STQAV,ALKIAV,
           3XASS,XSSS,XESS,XISS,XVSS,XNSS,SBPSS,SBSSS,SUSS,
           4NONSS,NH3SS,NO3SS,NUSS,ALKSS,VARNTL)
      C*****************************************************************
            FACTC=STQAV/(STIAV*QAV(1))
            FACTN=NTQAV/(NTIAV*QAV(1))
            XASS=FACTC*XASS
            XSSS=FACTC*XSSS
            XESS=FACTC*XESS
            XISS=FACTC*XISS
            XVSS=FACTC*XVSS
            XNSS=FACTN*XNSS
            CRITXV=CRIT*XVSS
            IF(CRITXV.LT.CRIT) CRITXV=CRIT
            CRITXS=CRIT*XSSS
            IF(CRITXS.LT.CRIT) CRITXS=CRIT
            CRISBP=CRIT*SBPSS
            IF(CRISBP.LT.CRIT) CRISBP=CRIT
            CRINO3=CRIT*NO3SS
            IF(CRINO3.LT.CRIT) CRINO3=CRIT
            CRITXN=CRIT*XNSS
            IF(CRITXN.LT.CRIT) CRITXN=CRIT
      C
      C
      C           CALCULATE INTEGRATION STEP LENGTHS
      C           ----------------------------------
      C
            DT=1./D
            DDT=1./DD
            RDT=DDT/DT
      C*****************************************************************
            CALL DSTRCY(CRIT,VP,XVSS,NR,LINI,LSRI,LARI,LARO,LBRI,LBRO,
           1QSR,QAR,QBR,SR,AR,BR,DT,DDT,RDT,NLL)
      C*****************************************************************
      C
```

```
C           RELATE STEP INTERVALS TO THE TIME BASE
C           --------------------------------------
C
      II=0
      KN=1440/D
      KH1=KH+1
      DO 27 I=1,NL,1
      JE=I-1
      IF(JE.EQ.0) GO TO 3
      KO(I)=KO(I-1)+KN
      GO TO 4
    3 KO(I)=KOM
    4 II=II+1
      IF (II.EQ.KH1) KO(I)=KO(I)+40
      IF (II.EQ.KH1) II=1
      IF (KO(I).EQ.2400) KO(I)=0
      IF(KO(I).EQ.ICSD) IC=I-1
      IF(KOM.EQ.ICSD) IC=1
      IF(KO(I).EQ.ISSD) IS=I
   27 CONTINUE
      HK=IS-IC
      DO 9 J=1,NR,1
      DO 29 I=1,NL,1
      WQ(J,I)=0.0
   29 CONTINUE
      IF(LRSD(J).EQ.0) GO TO 32
      DO 28 I=IC,IS,1
      WQ(J,I)=VP/RS*D/HK*1.0/NRSD/DSFCTR(J)
   28 CONTINUE
   32 DO 48 I=1,NL,1
      Q(J+1,I)=Q(J,I)-WQ(J,I)
      IF(Q(J+1,I).GT.0.00) GO TO 48
      Q(J+1,I)=0.00
      WQ(J,I)=Q(J,I)
   48 CONTINUE
    9 CONTINUE
C
C           SET INITIAL CONDITIONS
C           ----------------------
C
      DUM=(1.0+SR)/SR
      DO 7  J=1,NR,1
      DO 7 I=1,NL,1
      SBP(J,I)=SBPSS*DSFCTR(J)
      SBS(J,I)=SBSSS
      XS(J,I)=XSSS*DSFCTR(J)
      XA(J,I)=XASS*DSFCTR(J)
      XE(J,I)=XESS*DSFCTR(J)
      XI(J,I)=XISS*DSFCTR(J)
      XV(J,I)=XVSS*DSFCTR(J)
      SU(J,I)=SUSS
      NU(J,I)=NUSS
      NON(J,I)=NONSS
      NH3(J,I)=NH3SS
      XN(J,I)=XNSS*DSFCTR(J)
      NO3(J,I)=NO3SS
      ALK(J,I)=ALKSS
      IF(J.NE.NR) GO TO 7
      SBPR(I)=SBP(J,I)*DUM
      XSR(I)=XS(J,I)*DUM
      XAR(I)=XA(J,I)*DUM
      XER(I)=XE(J,I)*DUM
      XIR(I)=XI(J,I)*DUM
      XNR(I)=XN(J,I)*DUM
```

```
832           ALK(J,1)=ALK(J,NL)
833           XV(J,1)=XV(J,NL)
834         6 CONTINUE
835           GO TO 17
836        15 CONTINUE
837           DO 26 J=1,NR,1
838           DO 26 I=1,NL
839           JA=I-1
840           IF (JA.EQ.0) JA=D
841           JB=I+1
842           IF (JB.EQ.NLL) JB=2
843           JD=I
844           IF(JD.EQ.NL) JD=1
845           RXS(J,I)=(XS(J,JB)-XS(J,JA))/(DT*2.0)
846           ST(J,I)=SU(J,I)+SBS(J,I)
847           NT(J,I)=NON(J,I)+NU(J,I)+NH3(J,I)
848           OC(J,I)=(1.0-P*YH)*SYNTH(J,JD)/(DT*24.0)
849           ANPOT(J,I)=ANPOT(J,JD)/(DT*24.0)
850           ON(J,I)=4.57*NIT(J,JD)/(DT*24.0)
851           OT(J,I)=OC(J,I)+ON(J,I)
852           AMON(J,I)=AMON(J,JD)/(DT*24.0)
853        26 CONTINUE
854     C
855     C       CALCULATION OF AVERAGE RESULTS FROM DYNAMIC MODEL
856     C       ------------------------------------------------
857     C           EXAMPLE:  XVAV = AVERAGE XV
858     C
859           TRN=0.0
860           SUMXVT=0.0
861           SUMWST=0.0
862           SUMXAM=0.0
863           SUMXEM=0.0
864           SUMXIM=0.0
865           SUMXSM=0.0
866           SUMXNM=0.0
867           SUMODM=0.0
868           SMAPTM=0.0
869           APTMNR=0.0
870           RNO3AV=0.0
871           SUMOCM=0.0
872           SUMONM=0.0
873           SUMOTM=0.0
874           SUMAMM=0.0
875           SUNSWM=0.0
876           TWCOD=0.0
877           TWN=0.0
878           EN=0.0
879           ENH3=0.0
880           ENON=0.0
881           ENO3=0.0
882           EALK=0.0
883           ENT=0.0
884           ECOD=0.0
885           LNAME(1)='AROBIC'
886           LNAME(2)='ANOXIC'
887           DO 31 J=2,NRR,1
888           SUMQ=0.0
889           DO 30 I=1,D,1
890        30 SUMQ=SUMQ+Q(J,I)
891        31 QAV(J)=SUMQ/D
892           DO 22 J=1,NR,1
893           SUMWS=0.0
894           SUMNSW=0.0
895           SUMNU=0.0
```

```
      SUMXA=0.0
      SUMXE=0.0
      SUMXI=0.0
      SUMSBP=0.0
      SUMSBS=0.0
      SUMXV=0.0
      SUMXS=0.0
      SUMXN=0.0
      SUMOC=0.0
      SUMOD=0.0
      SUMAPT=0.0
      SUMON=0.0
      SUMSU=0.0
      SUMRXS=0.0
      SUMNON=0.0
      SUMNH3=0.0
      SUMNT=0.0
      SUMNO3=0.0
      SUMALK=0.0
      SUMOT=0.0
      SUMAM=0.0
      WCOD=0.0
      WN=0.0
      WALK=0.0
      TWALK=0.0
      DO 21 I=1,D
      SUMWS=SUMWS+XV(J,I)*WQ(J,I)*DT
      SUMNSW=SUMNSW+(FN*(XA(J,I)+XE(J,I)+XI(J,I))+FNS*XS(J,I))
     1*WQ(J,I)*DT
      IF(J.NE.NR) GO TO 11
      ECOD=ECOD+Q(J+1,I)*ST(J,I)*DT/QAV(J+1)
      ESBS=ESBS+Q(J+1,I)*(SBS(J,I))*DT/QAV(J+1)
      ESU=ESU+Q(J+1,I)*(SU(J,I))*DT/QAV(J+1)
      EN=EN+Q(J+1,I)*(NT(J,I)+NO3(J,I))*DT/QAV(J+1)
      ENH3=ENH3+Q(J+1,I)*(NH3(J,I))*DT/QAV(J+1)
      ENON=ENON+Q(J+1,I)*(NON(J,I))*DT/QAV(J+1)
      ENO3=ENO3+Q(J+1,I)*(NO3(J,I))*DT/QAV(J+1)
      EALK=EALK+Q(J+1,I)*(ALK(J,I))*DT/QAV(J+1)
      ENT=ENT+Q(J+1,I)*(NT(J,I))*DT/QAV(J+1)
   11 SUMNU=SUMNU+NU(J,I)
      SUMXA=SUMXA+XA(J,I)
      SUMXE=SUMXE+XE(J,I)
      SUMXI=SUMXI+XI(J,I)
      SUMSBP=SUMSBP+SBP(J,I)
      SUMSBS=SUMSBS+SBS(J,I)
      SUMXV=SUMXV+XV(J,I)
      SUMXN=SUMXN+XN(J,I)
      SUMXS=SUMXS+XS(J,I)
      SUMOC=SUMOC+OC(J,I)
      SUMAPT=SUMAPT+ANPOT(J,I)
      IF(NTDN(J).EQ.1) GO TO 12
      SUMOD=SUMOD+OC(J,I)
   12 SUMON=SUMON+ON(J,I)
      SUMSU=SUMSU+SU(J,I)
      SUMRXS=SUMRXS+RXS(J,I)
      SUMNO3=SUMNO3+NO3(J,I)
      SUMALK=SUMALK+ALK(J,I)
      SUMNON=SUMNON+NON(J,I)
      SUMNH3=SUMNH3+NH3(J,I)
      SUMNT=SUMNT+NT(J,I)
      WCOD=WCOD+WQ(J,I)*(ST(J,I)+SBP(J,I))*DT
      WN=WN+WQ(J,I)*(NT(J,I)+NO3(J,I))*DT
      WALK=WALK+WQ(J,I)*ALK(J,I)*DT
      SUMOT=SUMOT+OT(J,I)
```

```
      SUMAM=SUMAM+AMON(J,I)
   21 CONTINUE
      NUAV(J)=SUMNU/D
      XAAV(J)=SUMXA/D
      XEAV(J)=SUMXE/D
      XIAV(J)=SUMXI/D
      SBPAV(J)=SUMSBP/D
      SBSAV(J)=SUMSBS/D
      XVAV(J)=SUMXV/D
      XSAV(J)=SUMXS/D
      XNAV(J)=SUMXN/D
      OCAV(J)=SUMOC/D
      ODAV(J)=SUMOD/D
      APTAV(J)=SUMAPT/D
      APTNR(J)=APTAV(J)*24.0*V(J)/QAV(1)
      RNO3(J)=ODAV(J)*24.0*V(J)/(QAV(1)*2.85)
      ONAV(J)=SUMON/D
      SUAV(J)=SUMSU/D
      RXSAV(J)=SUMRXS/D
      STAV(J)=SUAV(J)+SBSAV(J)
      NSWAV(J)=SUMNSW/QAV(1)
      SUNSWM=SUNSWM+NSWAV(J)
      OTAV(J)=SUMOT/D
      AMAV(J)=SUMAM/D
      NO3AV(J)=SUMNO3/D
      ALKAV(J)=SUMALK/D
      NONAV(J)=SUMNON/D
      NH3AV(J)=SUMNH3/D
      NTAV(J)=SUMNT/D
      SUMWST=SUMWST+SUMWS
      SUMXVT=SUMXVT+XVAV(J)*V(J)
      SUMXAM=SUMXAM+XAAV(J)*V(J)
      SUMXEM=SUMXEM+XEAV(J)*V(J)
      SUMXIM=SUMXIM+XIAV(J)*V(J)
      SUMXSM=SUMXSM+XSAV(J)*V(J)
      SUMXNM=SUMXNM+XNAV(J)*V(J)
      SUMODM=SUMODM+ODAV(J)*V(J)
      SMAPTM=SMAPTM+APTAV(J)*V(J)
      APTMNR=APTMNR+APTNR(J)
      RNO3AV=RNO3AV+RNO3(J)
      SUMOCM=SUMOCM+OCAV(J)*V(J)
      SUMONM=SUMONM+ONAV(J)*V(J)
      SUMOTM=SUMOTM+OTAV(J)*V(J)
      SUMAMM=SUMAMM+AMAV(J)*V(J)
      I=NLL
      Q(J,I)=QAV(J)
C*****************************************************************
      CALL DUMVCY(J,I,DUMIF,DUMS,DUMA,DUMB,DUMIR,DUMO,
     1LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,
     1DT,DDT,RDT)
C*****************************************************************
      RA(J)=24.0*DDT/DUMO
      RN(J)=V(J)/QAV(J)*24.0
      TWCOD=TWCOD+WCOD
      TWN=TWN+WN
      TWALK=TWALK+WALK
      TRN=TRN+RN(J)
      JJ=NTDN(J)
      NAME(J)=LNAME(JJ)
      IF(NTDN(J).EQ.1) GO TO 22
      SUMXVA=SUMXVA+XVAV(J)*V(J)
   22 CONTINUE
      RSAV=SUMXVT/SUMWST
      XAMAV=SUMXAM/VP
```

```
1024            XEMAV=SUMXEM/VP
1025            X1MAV=SUMX1M/VP
1026            XSMAV=SUMXSM/VP
1027            XVMAV=SUMXVT/VP
1028            XNMAV=SUMXNM/VP
1029            ODMAV=SUMODM/VP
1030            APTMAV=SMAPTM/VP
1031            FANOX=SUMXVA/SUMXVT
1032            OCMAV=SUMOCM/VP
1033            ONMAV=SUMONM/VP
1034            AMMAV=SUMAMM/VP
1035      C
1036      C       CHECK MASS BALANCES ON COD , NITROGEN AND ALKALINITY
1037      C       ---------------------------------------
1038      C  1.  COD
1039      C  -------
1040            TCOD1=STQAV
1041            TCODO=P*SUMWST+SUMOCM*24.0+TWCOD+ECOD
1042            BALC=TCODO*100.0/TCOD1
1043      C  2.  NITROGEN
1044      C  ------------
1045            TNT1=NTQAV
1046            TNTO=SUNSWM*QAV(1)+TWN+EN+SUMODM*24/2.85
1047            BALN=TNTO*100.0/TNT1
1048      C  3.  ALKALINITY
1049      C  ---------------
1050            TALK1=ALKQAV
1051            TALKO=TWALK+EALK
1052           &+3.57*24.0*(2.0*SUMONM/4.57-SUMODM/2.86-SUMAMM)
1053            BALALK=TALKO/TALK1*100.0
1054      C
1055      C       OUTPUT AND FORMAT STATEMENTS
1056      C       ----------------------------
1057      C
1058            IF(RESULT.EQ.2) GO TO 126
1059            WRITE(5,201)TEMP,PH,YH,BHT,KVT,YN,THEN,F,FCA,FN,KNT,PHIN,
1060           1THEE,P,FOE,UMT,BNT,FOS,KRT,KSPT,THEZ,KMPT,THEA,THES,
1061           2KMST,KSST,FR,FNS,RSM,RS,RSA
1062         201 FORMAT(1H1,20X,22(1H*),/,1H ,20X,22(1H*),/,
1063           11H ,20X,22(1H*),/,
1064           11H0,10X,'KINETIC CONSTANTS',/,1H ,10X,17(1H-),/,
1065           21H ,5X,'PROCESS OPERATING TEMPERATURE=',F5.1,1X,'DEG. C',/,
1066           31H ,28X,'AND PH=',F5.2,/,1H0,5X,'HETEROTROPHS',28X,'AUTOTROPHS',/,
1067           41H ,5X,12(1H-),28X,10(1H-),/,1H ,6X,'YH=',F5.3,4X,'BH=',F5.3,4X,
1068           5'KV=',F5.3,8X,'YN=',F5.3,6X,'THEN=',F5.3,/,1H ,7X,'F=',F5.3,3X,
1069           6'FCA=',F5.3,4X,'FN=',F5.3,8X,'KN=',F5.2,6X,'PHIN=',F5.3,/,
1070           75X,'THEE=',F5.3,5X,'P=',F5.3,3X,'FOE=',F5.3,8X,'UM=',F5.3,
1071           28X,'BN=',F5.3,/,
1072           86X,'FOS=',F5.3,4X,'KR=',F5.3,3X,'KSP=',F5.1,6X,'THEZ=',F5.3,/,
1073           16X,'KMP=',F5.3,2X,'THEA=',F5.3,2X,'THES=',F5.3,/,
1074           26X,'KMS=',F5.2,3X,'KSS=',F5.1,/,
1075           27X,'FR=',F5.3,3X,'FNS=',F5.3,/,
1076           91H0,5X,'MINIMUM SLUDGE AGE FOR NITRIFICATION=',F5.2,1X,'DAYS',/,
1077           11H ,5X,'OPERATING SLUDGE AGE OF PROCESS=',F5.2,1X,'DAYS',/,
1078           11H ,5X,'AEROBIC SLUDGE AGE OF PROCESS=',F5.2,1X,'DAYS')
1079            WRITE(5,202)FUP,FUS,SN,UN,QAV(1),STQAV,NTQAV,STIAV,SBIAV,SUIAV,
1080           1XIIAV,NTIAV,NH3IAV,NONIAV,NUIAV,NO3IAV,ALKIAV
1081         202 FORMAT(1H0,10X,'AVERAGE COMPOSITION OF SEWAGE',/,
1082           11H ,10X,29(1H-),/,1H0,5X,'SOLID INERT FRACTION OF INFLUENT COD=',
1083           2F5.3,/,1H ,5X,'SOLUBLE INERT FRACTION OF INFLUENT COD=',F5.3,/,
1084           31H ,5X,'AMMONIA TO TKN FRACTION=',F5.3,/,
1085           41H ,5X,'INERT TKN FRACTION=',F5.3,/,1H0,4X,'QAV=',F9.5,
1086           15X,'STQAV=',F9.5,5X,'NTQAV=',F9.5,/,
1087           51H ,5X,'STI=',F7.2,5X,'SBI=',F7.2,4X,'SUI=',F6.2,
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1088          64X,'XII=',F6.2,/,1H ,5X,'NTI=',F6.2,4X,'NH3I=',F6.2,
1089          74X,'NONI=',F5.2,5X,'NUI=',F5.2,5X,'NO3I=',F5.2,/,
1090          &4X,'ALKI=',F7.2)
1091           IF(TYPE.EQ.1) GO TO 122
1092           IF(TYPE.EQ.2) GO TO 123
1093           IF(TYPE.EQ.3) GO TO 122
1094        122 WRITE(5,205)VARQ,VARSTI,VARNTI,VNO3I,VALKI,VARSTL,VARNTL,VRNO3L,
1095          &VRALKL
1096        205 FORMAT(1H0,10X,'SINE WAVE INFLUENT FLOW AND FEED PATTERN',/,
1097          11H ,10X,40(1H-),/,
1098          41H0,10X,'AMPLITUDE OF FLOW WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1099          51H ,10X,'AMPLITUDE OF COD WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1100          61H ,10X,'AMPLITUDE OF TKN WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1101          71H ,10X,'AMPLITUDE OF NO3 WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1102          71H ,10X,'AMPLITUDE OF ALK WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1103          81H ,10X,'AMPLITUDE OF COD LOAD WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1104          81H ,10X,'AMPLITUDE OF TKN LOAD WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1105          81H ,10X,'AMPLITUDE OF NO3 LOAD WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1106          81H ,10X,'AMPLITUDE OF ALK LOAD WAVE=',F5.2,4X,'(FRAC OF AVE)')
1107           WRITE(5,204) NR,LINI,LSRI,LARO,LARI,LBRO,LBRI,SR,AR,BR
1108        204 FORMAT(1H0,10X,'PROCESS CONFIGURATION DATA',/,1H ,5X,26(1H-),/,
1109          11H0,5X,'NUMBER OF TANKS IN SERIES',I2,/,
1110          11H ,5X,'INFLUENT FLOW INTO TANK',I2,/,
1111          11H ,5X,'S-RECYCLE FROM SETTLER TO TANK ',I2,/,
1112          11H ,5X,'A-RECYCLE FROM TANK',I2,1X,'INTO TANK',I2,/,
1113          11H ,5X,'B-RECYCLE FROM TANK',I2,1X,'INTO TANK',I2,/,
1114          11H ,5X,'S RECYCLE RATIO=',F4.1,/,
1115          11H ,5X,'A RECYCLE RATIO=',F4.1,/,
1116          11H ,5X,'B RECYCLE RATIO=',F4.1)
1117           GO TO 124
1118        123 WRITE(5,206)LFP,PQ,BQ,VF
1119        206 FORMAT(1H0,10X,'SQUARE WAVE INFLUENT FLOW AND FEED PATTERN',/,
1120          11H ,10X,42(1H-),/,
1121          11H ,10X,'LENGTH OF FEED PERIOD (HRS)=',F5.2,/,
1122          21H ,10X,'PEAK FLOW (L/D)=',F9.5,/,
1123          31H ,10X,'BASE FLOW (L/D)=',F9.5,/,
1124          41H ,10X,'VOLUME OF FEED PER DAY (LITRES)=' F9.5)
1125        124 WRITE(5,203) TRN,VP,VPA,FANOX,M,RSAV
1126        203 FORMAT(1H1,10X,21(1H-),/,1H ,10X,'DAILY AVERAGE RESULTS',/,
1127          11H ,10X,21(1H-),/,
1128          11H0,5X,'TOTAL HYDRAULIC RETENTION TIME=',F8.3,1X,'HOURS',/,
1129          11H ,5X,'TOTAL VOLUME=',F9.5,1X,'LITRES',/,
1130          11H ,5X,'AEROBIC VOLUME=',F9.5,1X,'LITRES',/,
1131          11H ,5X,'ANOXIC FRACTION=',F5.3,/,
1132          21H ,5X,'NO. OF DAYS TO REACH DYNAMIC STEADY STATE=',I3,/,
1133          21H ,5X,'SLUDGE AGE =',F5.2,1X,'DAYS')
1134           WRITE(5,244)TCODI,TCODO,BALC,TNTI,TNTO,BALN,TALKI,TALKO,BALALK
1135        244 FORMAT(1H0,10X,'MASS BALANCES ON COD AND NITROGEN',/,
1136          11H ,10X,33(1H-),/,
1137          21H0,5X,'TOTAL COD MASS INPUT=',1X,E13.6,1X,'(MG-COD/D)',/,
1138          21H ,5X,'TOTAL COD MASS OUTPUT=',1X,E13.6,1X,'(MG-COD/D)',/,
1139          31H ,5X,'PERCENTAGE COD RECOVERY=',1X,F7.3,1X,'(%)',/,
1140          21H ,5X,'TOTAL NITROGEN MASS INPUT=',1X,E13.6,1X,'(MG-N/D)',/,
1141          21H ,5X,'TOTAL NITROGEN MASS OUTPUT=',1X,E13.6,1X,'(MG-N/D)',/,
1142          31H ,5X,'PERCENTAGE NITROGEN RECOVERY=',1X,F7.3,1X,'(%)',/,
1143          21H ,5X,'TOTAL ALKALINITY MASS INPUT=',1X,E13.6,1X,'(CACO3/D)',/,
1144          21H ,5X,'TOTAL ALKALINITY MASS OUTPUT=',1X,E13.6,1X,'(CACO3/D)',/,
1145          31H ,5X,'PERCENTAGE ALKALINITY RECOVERY=',1X,F7.3,1X,'(%)')
1146           WRITE(5,300)
1147        300 FORMAT(1H0,30X,'TANK  1',6X,'TANK  2',
1148          26X,'TANK  3',6X,'TANK  4',6X,'TANK  5',6X,'TANK  6',
1149          36X,'MEAN',/,1H ,30X,7(1H-),6X,7(1H-),6X,7(1H-),6X,7(1H-),
1150          46X,7(1H-),6X,7(1H-),6X,4(1H-))
1151           WRITE(5,327) (NAME(J),J=1,NR,1)
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1152        327 FORMAT(1H ,23X,6A13)
1153            WRITE(5,328)
1154        328 FORMAT(1H ,30X,6(1H-),7X,6(1H-),7X,6(1H-),
1155           17X,6(1H-),7X,6(1H-),7X,6(1H-))
1156            WRITE(5,301) (XAAV(J),J=1,NR,1),XAMAV
1157            WRITE(5,302) (XSAV(J),J=1,NR,1),XSMAV
1158            WRITE(5,303) (XEAV(J),J=1,NR,1),XEMAV
1159            WRITE(5,304) (XIAV(J),J=1,NR,1),XIMAV
1160            WRITE(5,305) (XVAV(J),J=1,NR,1),XVMAV
1161            WRITE(5,319) (XNAV(J),J=1,NR,1),XNMAV
1162            WRITE(5,313) (ODAV(J),J=1,NR,1),ODMAV
1163            WRITE(5,314) (OCAV(J),J=1,NR,1),OCMAV
1164            WRITE(5,315) (ONAV(J),J=1,NR,1),ONMAV
1165            WRITE(5,316) (OTAV(J),J=1,NR,1),OTMAV
1166            WRITE(5,3160) (AMAV(J),J=1,NR,1),AMMAV
1167            WRITE(5,309) (NSWAV(J),J=1,NR,1),SUNSWM
1168            WRITE(5,321) (SBPAV(J),J=1,NR,1)
1169            WRITE(5,320) (SBSAV(J),J=1,NR,1)
1170            WRITE(5,324) (SUAV(J),J=1,NR,1)
1171            WRITE(5,311) (STAV(J),J=1,NR,1)
1172            WRITE(5,306) (NONAV(J),J=1,NR,1)
1173            WRITE(5,310) (NH3AV(J),J=1,NR,1)
1174            WRITE(5,323) (NUAV(J),J=1,NR,1)
1175            WRITE(5,312) (NTAV(J),J=1,NR,1)
1176            WRITE(5,307) (NO3AV(J),J=1,NR,1)
1177            WRITE(5,703) (ALKAV(J),J=1,NR,1)
1178            WRITE(5,308) (RNO3(J),J=1,NR,1),RNO3AV
1179            WRITE(5,326) (APTAV(J),J=1,NR,1),APTMAV
1180            WRITE(5,329) (APTNR(J),J=1,NR,1),APTMNR
1181            WRITE(5,322) (V(J),J=1,NR,1)
1182            WRITE(5,325) (QAV(J),J=1,NRR,1)
1183            WRITE(5,317) (RA(J),J=1,NR,1)
1184            WRITE(5,318) (RN(J),J=1,NR,1)
1185        301 FORMAT(1H ,8X,'XA    (MG-VSS/L)',2X,7(E13.5))
1186        302 FORMAT(1H ,8X,'XS    (MG-VSS/L)',2X,7(E13.5))
1187        303 FORMAT(1H ,8X,'XE    (MG-VSS/L)',2X,7(E13.5))
1188        304 FORMAT(1H ,8X,'XI    (MG-VSS/L)',2X,7(E13.5))
1189        305 FORMAT(1H ,8X,'XV    (MG-VSS/L)',2X,7(E13.5))
1190        319 FORMAT(1H ,8X,'XN    (MG-VSS/L)',2X,7(E13.5))
1191        313 FORMAT(1H ,8X,'OD  (MG-O/L/HR)',2X,7(E13.5))
1192        314 FORMAT(1H ,8X,'OC  (MG-O/L/HR)',2X,7(E13.5))
1193        315 FORMAT(1H ,8X,'ON  (MG-O/L/HR)',2X,7(E13.5))
1194        316 FORMAT(1H ,8X,'OT  (MG-O/L/HR)',2X,7(E13.5))
1195       3160 FORMAT(1H ,8X,'AM  (MG-N/L/HR)',2X,7(E13.5))
1196        309 FORMAT(1H ,8X,'NSW  (MG-N/L/D)',2X,7(E13.5))
1197        321 FORMAT(1H ,8X,'SBP  (MG-COD/L)',2X,6(E13.5))
1198        320 FORMAT(1H ,8X,'SBS  (MG-COD/L)',2X,6(E13.5))
1199        324 FORMAT(1H ,8X,'SU   (MG-COD/L)',2X,6(E13.5))
1200        311 FORMAT(1H ,8X,'ST   (MG-COD/L)',2X,6(E13.5))
1201        306 FORMAT(1H ,8X,'NON    (MG-N/L)',2X,6(E13.5))
1202        310 FORMAT(1H ,8X,'NH3    (MG-N/L)',2X,6(E13.5))
1203        323 FORMAT(1H ,8X,'NU     (MG-N/L)',2X,6(E13.5))
1204        312 FORMAT(1H ,8X,'NT     (MG-N/L)',2X,6(E13.5))
1205        307 FORMAT(1H ,8X,'NO3    (MG-N/L)',2X,6(E13.5))
1206        703 FORMAT(1H ,8X,'ALK (MG-CACO3/L)',2X,6(E13.5))
1207        326 FORMAT(1H ,8X,'APT (MG-N/L/HR)',2X,7(E13.5))
1208        329 FORMAT(1H ,8X,'APTNR  (MG-N/L)',2X,7(E13.5))
1209        308 FORMAT(1H ,8X,'RNO3   (MG-N/L)',2X,7(E13.5))
1210        322 FORMAT(1H ,8X,'VOL    (LITRES)',2X,6(E13.5))
1211        325 FORMAT(1H ,8X,'FLOW      (L/D)',2X,7(E13.5))
1212        317 FORMAT(1H ,8X,'AHRT    (HOURS)',2X,6(E13.5))
1213        318 FORMAT(1H ,8X,'NHRT    (HOURS)',2X,6(E13.5))
1214            WRITE(5,3000)ECOD,ESBS,ESU,EN,ENT,ENH3,ENON,ENO3,EALK
1215       3000 FORMAT(1HO,8X,'AVERAGE EFFLUENT CONCENTRATIONS',/,
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1216          &8X,'TOT-COD=',E13.5,/,
1217          &8X,'SBS',E13.5,/,
1218          &8X,'SU',E13.5,/,
1219          &8X,'TOTAL-N=',E13.5,/,
1220          &8X,'TKN',E13.5,/,
1221          &8X,'NH3',E13.5,/,
1222          &8X,'NON',E13.5,/,
1223          &8X,'NO3',E13.5,/,
1224          &8X,'ALK',E13.5)
1225           WRITE(CH,'(I1)')NR
1226           IMFT(10:10)=CH
1227           WRITE(5,399) (KO(1),STQ(I),NTQ(I),Q(1,I),SBI(I),STI(I),
1228          1NTI(I),SUI(I),NUI(I),XII(I),NH3I(I),NONI(I),I=1,NL,KH)
1229       399 FORMAT(1H1,10X,'DYNAMIC INPUTS',/,1H ,10X,14(1H-),/,
1230          11H ,2X,'TIME',2X,'COD LOAD',2X,'TKN LOAD',4X,'FLOW',7X,'SBI',7X,
1231          2'STI',7X,'NTI',7X,'SUI',7X,'NUI',7X,'XII',6X,'NH3I',6X,'NONI',/,
1232          31H ,2X,4(1H-),2X,8(1H-),3X,8(1H-),4X,4(1H-),7X,3(1H-),7X,3(1H-),
1233          47X,3(1H-),7X,3(1H-),7X,3(1H-),7X,3(1H-),6X,4(1H-),6X,4(1H-),/,
1234          5(1H ,2X,I4,11(E10.4)))
1235           WRITE(5,400)
1236       400 FORMAT(1H1,10X,21(1H-),/,1H ,10X,'DAILY DYNAMIC RESULTS',/,
1237          11H ,10X,21(1H-),/,
1238          11H0,20X,'TIME',5X,'TANK  1',6X,'TANK  2',
1239          26X,'TANK  3',6X,'TANK  4',6X,'TANK  5',6X,'TANK  6',/,
1240          31H ,20X,4(1H-),5X,7(1H-),6X,7(1H-),6X,7(1H-),6X,7(1H-),
1241          46X,7(1H-),6X,7(1H-))
1242           WRITE(5,401)
1243       401 FORMAT(1H0,3X,'XA    (MG-VSS/L)')
1244           WRITE(5,IMFT) (KO(I),(XA(J,I),J=1,NR,1),I=1,NL,KH)
1245           WRITE(5,402)
1246       402 FORMAT(1H0,3X,'XS    (MG-VSS/L)')
1247           WRITE(5,IMFT) (KO(I),(XS(J,I),J=1,NR,1),I=1,NL,KH)
1248           WRITE(5,403)
1249       403 FORMAT(1H0,3X,'XE    (MG-VSS/L)')
1250           WRITE(5,IMFT) (KO(I),(XE(J,I),J=1,NR,1),I=1,NL,KH)
1251           WRITE(5,404)
1252       404 FORMAT(1H0,3X,'XI    (MG-VSS/L)')
1253           WRITE(5,IMFT) (KO(I),(XI(J,I),J=1,NR,1),I=1,NL,KH)
1254           WRITE(5,405)
1255       405 FORMAT(1H0,3X,'XV    (MG-VSS/L)')
1256           WRITE(5,IMFT) (KO(I),(XV(J,I),J=1,NR,1),I=1,NL,KH)
1257           WRITE(5,419)
1258       419 FORMAT(1H0,3X,'XN    (MG-VSS/L)')
1259           WRITE(5,IMFT) (KO(I),(XN(J,I),J=1,NR,1),I=1,NL,KH)
1260           WRITE(5,407)
1261       407 FORMAT(1H0,3X,'OC  (MG-O/L/HR)')
1262           WRITE(5,IMFT) (KO(I),(OC(J,I),J=1,NR,1),I=1,NL,KH)
1263           WRITE(5,408)
1264       408 FORMAT(1H0,3X,'ON  (MG-O/L/HR)')
1265           WRITE(5,IMFT) (KO(I),(ON(J,I),J=1,NR,1),I=1,NL,KH)
1266           WRITE(5,409)
1267       409 FORMAT(1H0,3X,'OT  (MG-O/L/HR)')
1268           WRITE(5,IMFT) (KO(I),(OT(J,I),J=1,NR,1),I=1,NL,KH)
1269      4090  FORMAT(1H0,3X,'AM (MG-N/L/HR)')
1270           WRITE(5,IMFT) (KO(I),(AMON(J,I),J=1,NR,1),I=1,NL,KH)
1271           WRITE(5,417)
1272       417 FORMAT(1H0,3X,'WQ       (L/DAY)')
1273           WRITE(5,IMFT) (KO(I),(WQ(J,I),J=1,NR,1),I=1,NL,KH)
1274           WRITE(5,411)
1275       411 FORMAT(1H0,3X,'SBP   (MG-COD/L)')
1276           WRITE(5,IMFT) (KO(I),(SBP(J,I),J=1,NR,1),I=1,NL,KH)
1277           WRITE(5,418)
1278       418 FORMAT(1H0,3X,'SBS   (MG-COD/L)')
1279           WRITE(5,IMFT) (KO(I),(SBS(J,I),J=1,NR,1),I=1,NL,KH)
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1280            WRITE(5,420)
1281        420 FORMAT(1HO,3X,'SU    (MG-COD/L)')
1282            WRITE(5,IMFT) (KO(I),(SU(J,I),J=1,NR,1),I=1,NL,KH)
1283            WRITE(5,412)
1284        412 FORMAT(1HO,3X,'ST    (MG-COD/L)')
1285            WRITE(5,IMFT) (KO(I),(ST(J,I),J=1,NR,1),I=1,NL,KH)
1286            WRITE(5,413)
1287        413 FORMAT(1HO,3X,'NON     (MG-N/L)')
1288            WRITE(5,IMFT) (KO(I),(NON(J,1),J=1,NR,1),I=1,NL,KH)
1289            WRITE(5,421)
1290        421 FORMAT(1HO,3X,'NU      (MG-N/L)')
1291            WRITE(5,IMFT) (KO(I),(NU(J,I),J=1,NR,1),I=1,NL,KH)
1292            WRITE(5,415)
1293        415 FORMAT(1HO,3X,'NH3     (MG-N/L)')
1294            WRITE(5,IMFT) (KO(I),(NH3(J,I),J=1,NR,1),I=1,NL,KH)
1295            WRITE(5,414)
1296        414 FORMAT(1HO,3X,'NT      (MG-N/L)')
1297            WRITE(5,IMFT) (KO(I),(NT(J,I),J=1,NR,1),I=1,NL,KH)
1298            WRITE(5,422)
1299        422 FORMAT(1HO,3X,'APT (MG-N/L/HR)')
1300            WRITE(5,IMFT) (KO(I),(ANPOT(J,I),J=1,NR,1),I=1,NL,KH)
1301            WRITE(5,416)
1302        416 FORMAT(1HO,3X,'NO3     (MG-N/L)')
1303            WRITE(5,IMFT) (KO(I),(NO3(J,I),J=1,NR,1),I=1,NL,KH)
1304            WRITE(5,461)
1305        461 FORMAT(1HO,3X,'ALK     (MG-CACO3/L)')
1306            WRITE(5,IMFT) (KO(I),(ALK(J,I),J=1,NR,1),I=1,NL,KH)
1307            IF(RESULT.NE.0) GO TO 127
1308        126 WRITE(18,522)ECOD,ENT,ENH3,ENO3,EALK
1309            WRITE(18,522)KOM,D,INIT(1),ANOX(1),AROB(1),IFREQ,NDH
1310        522 FORMAT()
1311            WRITE(18,45)(OT(1,I),XV(1,I),ST(1,I),NT(1,I),NH3(1,I),
1312           &NO3(1,I),ALK(1,I),I=1,NDH)
1313            WRITE(18,522) START,END
1314        45  FORMAT(7E13.5)
1315            ENDFILE 18
1316        127 STOP
1317            END
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      C*************************************************************
            SUBROUTINE PAINCY(YH,FCA,RS,RSA,BHT,KVT,TRN,KMPT,KSPT,KMST,KSST,
           1P,F,FN,FNS,KRT,KNT,BNT,UMT,YN,NITDEN,SR,AR,BR,FACT,FR,
           2SBIAV,SUIAV,XIIAV,NUIAV,NONIAV,NTIAV,NO3IAV,STQAV,ALKIAV,
           3XASS,XSSS,XESS,XISS,XVSS,XNSS,SBPSS,SBSSS,SUSS,
           4NONSS,NH3SS,NO3SS,NUSS,ALKSS,VARNTL)
      C*************************************************************
      C              THIS SUBROUTINE CALCULATES THE STEADY STATE
      C                SOLUTION ( PARTICULAR INTEGRAL )
      C*************************************************************
            PARAMETER NE=6
            PARAMETER NEE=7
            PARAMETER ND=241
            PARAMETER NDD=242
            REAL NUIAV,NONIAV,NTIAV,NO3IAV
            REAL NH3MAX,NSW
            COMMON/EXTC/SBS(NE,ND),SU(NE,ND),XA(NE,ND),XE(NE,ND),XI(NE,ND),
           1XV(NE,ND),XS(NE,ND),NU(NE,ND),NON(NE,ND),XN(NE,ND),NH3(NE,ND),
           2NO3(NE,ND),SBP(NE,ND),ALK(NE,ND)
            COMMON/BLOCKC/Q(NEE,NDD),V(NE),SBI(ND),SUI(ND),NIT(NE,ND),
           1NUI(ND),XII(ND),NONI(ND),NH3I(ND),NO3I(ND),SBPR(ND),ALKI(ND),
           2XAR(ND),XSR(ND),XER(ND),XIR(ND),XNR(ND),SYNTH(ND,ND),QAV(NEE),
           3ANPOT(NE,ND),AMON(NE,ND),DSPCTR(NE)
            REAL KVT,KMPT,KSPT,KMST,KSST,KRT,KNT,KMPTEF
            REAL NUI,NONI,NH3I,NO3I,NIT
            REAL NU,NON,NH3,NO3
            REAL NUSS,NONSS,NH3SS,NO3SS
            KMPTEF=KMPT*(FACT*(1.0-RSA/RS)+RSA/RS)
            XA0=YH*RS/(TRN*(1.+BHT*RS*(1.-P*YH*(1.-F))))*SBIAV
            SBSI=FCA*SBIAV
            SBPI=SBIAV-SBSI
            SBS0=0.0
            SBP0=0.0
            XS0=0.0
            DO 15 KLM=1,100,1
            C6=SBSI-KSST-TRN*KMST*XA0
            SBSS=(C6+(C6*C6+4.0*KSST*SBSI)**0.5)/2.0
            SBPSS=(SBPI/TRN+(1.0-F)*BHT*P*XA0)/(1.0/RS+KVT*XA0*(FR-XS0/XA0))
            C1=-(P/RS+KVT*SBPSS)/XA0
            C2=KVT*SBPSS*(P-KSPT)/P-KMPTEF-KSPT/RS
            C3=KVT*SBPSS*KSPT*XA0/P
            XSSS=(-C2-(C2*C2-4.0*C1*C3)**0.5)/(2.0*C1)
            SB=TRN/RS*(XSSS*P+SBPSS)+SBSS
            SYNEF=SB/SBIAV
            XASS=YH*(1.0-SYNEF)*SBIAV*RS/
           &(TRN*(1.0+BHT*RS*(1.0-P*YH*(1.0-F))))
            IF(XASS.LT.0.0)GO TO 55
            IF(XSSS.GE.(FR*XASS)) XSSS=FR*XASS
      4     DIF=(XS0-XSSS)**2.0
            DIF2=(SBP0-SBPSS)**2.0
            IF(DIF.LT.0.01.AND.DIF2.LT.0.01) GO TO 6
            SBS0=SBSS
            SBP0=SBPSS
            XS0=XSSS
            XA0=XASS
            IF (KLM.EQ.100) GO TO 55
      15    CONTINUE
      6     XESS=F*BHT*XASS*RS
            XISS=XIIAV*RS/TRN
            XVSS=XASS+XSSS+XESS+XISS
            SUSS=SUIAV
            NUSS=NUIAV
      C
```

```
64      25    FORMAT(1H, 5X,10F11.3,I4)
65            NSW=(FN*(XASS+XESS+XISS)+FNS*XSSS)*TRN/RS
66            NONSS=(NONIAV+TRN*((1.0-F)*FN*BNT*XASS-FNS*XSSS))/
67           1(1.0+KRT*XASS*TRN)
68            NH3MAX=NTIAV-NSW-NUSS-NONSS
69            NH3SS=KNT*(BNT+1.0/RS)/(UMT*RSA/RS-BNT-1.0/RS)*(1+VARNTL/TRN)
70            IF(NH3SS.GT.NH3MAX) NH3SS=NH3MAX
71            IF(NH3SS.LE.0.0) NH3SS=NH3MAX
72            XNSS=YN*(NH3MAX-NH3SS)*RS/(TRN*(1.0+BNT*RS))
73            NO3SS=NO3IAV+NH3MAX-NH3SS
74            IF(RS.EQ.RSA) GO TO 35
75            DENCON=((1-P*YH)/2.86)*KMPT*FACT*P*XSSS/(XSSS*P+KSPT*XASS)
76            DENCAP=((1-P*YH)/2.86)*FCA*SBIAV+DENCON*XASS*(1.0-RSA/RS)
77            NO3SS=NO3SS-DENCAP
78            ALKSS=ALKIAV-7.14*(NH3MAX-NH3SS)
79           &+3.57*(NONIAV-NONSS-NSW+DENCAP)
80            GO TO 35
81      55    WRITE(5,65)
82      65    FORMAT(1H ,10X,'NO SOLUTION AFTER 100 ITERATIONS OR XA.LT.0.0')
83      35    RETURN
84            END
```

```
      C*****************************************************************
            SUBROUTINE DSTRCY(CRIT,VP,XVSS,NR,LINI,LSRI,LARI,LARO,LBRI,LBRO,
           1QSR,QAR,QBR,SR,AR,BR,DT,DDT,RDT,NLL)
      C*****************************************************************
      C            THIS SUBROUTINE DISTRIBUTES THE SLUDGE MASS IN THE
      C             PROCESS TO EACH REACTOR IN ACCORDANCE WITH THE PROCESS
      C             CONFIGURATION INPUT DATA AND CALCULATES THE SLUDGE
      C             CONCENTRATION IN EACH REACTOR AS A FRACTION OF THE
      C             AVERAGE PROCESS SLUDGE CONCENTRATION
      C*****************************************************************
            PARAMETER NE=6
            PARAMETER NEE=7
            PARAMETER ND=241
            PARAMETER NDD=242
            DOUBLE PRECISION DT,DDT,RDT
            COMMON/EXTC/SBS(NE,ND),SU(NE,ND),XA(NE,ND),XE(NE,ND),XI(NE,ND),
           1XV(NE,ND),XS(NE,ND),NU(NE,ND),NON(NE,ND),XN(NE,ND),NH3(NE,ND),
           2NO3(NE,ND),SBP(NE,ND),ALK(NE,ND)
            COMMON/BLOCKC/Q(NEE,NDD),V(NE),SBI(ND),SUI(ND),NIT(NE,ND),
           1NUI(ND),XII(ND),NONI(ND),NH3I(ND),NO3I(ND),SBPR(ND),ALKI(ND),
           2XAR(ND),XSR(ND),XER(ND),XIR(ND),XNR(ND),SYNTH(NE,ND),QAV(NEE),
           3ANPOT(NE,ND),AMON(NE,ND),DSFCTR(NE)
            DIMENSION XBGN(NE),STXBGN(NE)
            CRITXB=CRIT*XVSS*0.001
            I=NLL
            DO 1 J=1,NR,1
            XBGN(J)=XVSS
            Q(J,I)=QAV(1)
          1 CONTINUE
            MM=0
          7 DO 2 J=1,NR,1
            STXBGN(J)=XBGN(J)
          2 CONTINUE
            DO 3 J=1,NR,1
            K=J-1
            IF(K.EQ.0) K=1
      C*****************************************************************
            CALL DUMVCY(J,I,DUMIF,DUMS,DUMA,DUMB,DUMIR,DUMO,
           1LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,
           1DT,DDT,RDT)
      C*****************************************************************
            XBGN(J)=XBGN(J)+DUMA*XBGN(LARO)+DUMB*XBGN(LBRO)
           1+DUMS*(1+SR)/SR*XBGN(NR)+DUMIR*XBGN(K)-DUMO*XBGN(J)
          3 CONTINUE
            DO 4 J=1,NR,1
            IF(ABS(XBGN(J)-STXBGN(J)).GE.CRITXB) GO TO 5
          4 CONTINUE
            GO TO 6
          5 MM=MM+1
            IF(MM.GE.5000) GO TO 6
            GO TO 7
          6 CONTINUE
            TXBGNO=0.0
            DO 9 J=1,NR,1
            TXBGNO=TXBGNO+V(J)*XBGN(J)
            DSFCTR(J)=XBGN(J)/XVSS
          9 CONTINUE
            XBAL=TXBGNO/(XVSS*VP)*100
            WRITE(5,104) MM,XVSS,XBAL
        104 FORMAT(1H0,5X,'NO. OF STEPS TO REACH STEADY STATE= ',I5,/,
           11H ,5X,'AVERAGE PROCESS SLUDGE CONCENTRATION= ',F6.1,/,
           11H ,5X,'PERCENTAGE SLUDGE RECOVERY = ',F7.3)
            WRITE(5,105) (DSFCTR(J),J=1,NR,1)
```

```
64          105 FORMAT((1H ,5X,F10.4))
65              RETURN
66              END
```

```
      C*****************************************************************
            SUBROUTINE ANOXCY(D,DT,J,FR,FCA,FCS,KVT,KMPT,KSPT,KMST,KSST,
           1P,NR,YH,BHT,F,KRT,FOE,FN,FOS,FNS,NJ,DDT,UMT,YN,KNT,RDT,BNT,
           2FACT,LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR)
      C*****************************************************************
      C             THIS SUBROUTINE CALCULATES THE DYNAMIC RESPONSE OF A
      C               REACTOR IN A SERIES UNDER ANOXIC CONDITIONS
      C*****************************************************************
            PARAMETER NE=6
            PARAMETER NEE=7
            PARAMETER ND=241
            PARAMETER NDD=242
            INTEGER D
            DOUBLE PRECISION DT,DDT,RDT
            COMMON/EXTC/SBS(NE,ND),SU(NE,ND),XA(NE,ND),XE(NE,ND),XI(NE,ND),
           1XV(NE,ND),XS(NE,ND),NU(NE,ND),NON(NE,ND),XN(NE,ND),NH3(NE,ND),
           2NO3(NE,ND),SBP(NE,ND),ALK(NE,ND)
            COMMON/BLOCKC/Q(NEE,NDD),V(NE),SBI(ND),SUI(ND),NIT(NE,ND),
           1NUI(ND),XII(ND),NONI(ND),NH3I(ND),NO3I(ND),SBPR(ND),ALKI(ND),
           2XAR(ND),XSR(ND),XER(ND),XIR(ND),XNR(ND),SYNTH(NE,ND),QAV(NEE),
           3ANPOT(NE,ND),AMON(NE,ND),DSFCTR(NE)
            REAL KVT,KMPT,KSPT,KMST,KSST,KRT,KNT
            REAL NUI,NONI,NH3I,NO3I,NIT
            REAL NU,NON,NH3,NO3
            REAL NH3A,NH3S,NH3T,NH3B,NH3K,NITT,NNIT,NO3A,NO3S,NO3T,NO3B,NO3K
            RNCOD=2.85/(1.0-P*YH)
            DO 11 I=1,D,1
      C*****************************************************************
            CALL DUMVCY(J,I,DUMF,DUMS,DUMA,DUMB,DUMR,DUMO,
           1LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,
           1DT,DDT,RDT)
      C*****************************************************************
            K=J-1
            IF(K.EQ.0) K=1
            SCER=P*(1.0-F)*BHT*XA(J,I)*DDT
            SCERN=FN*SCER/P
            CVNON=KRT*XA(J,I)*NON(J,I)*DDT
            SCVOD=KMPT*XS(J,I)/(KSPT*XA(J,I)+XS(J,I)*P)*FACT*XA(J,I)*DDT
            SCVO=0.00
            SCIS=0.00
            NNIT=0.00
            ANPOTS=0.00
            SBST=SBS(J,I)
            DFSBSS=(SBS(NR,I+1)-SBS(NR,I))/NJ
            DFSBSA=(SBS(LARO,I+1)-SBS(LARO,I))/NJ
            DFSBSB=(SBS(LBRO,I+1)-SBS(LBRO,I))/NJ
            DFSBSK=(SBS(K,I+1)-SBS(K,I))/NJ
            NH3T=NH3(J,I)
            DFNH3S=(NH3(NR,I+1)-NH3(NR,I))/NJ
            DFNH3A=(NH3(LARO,I+1)-NH3(LARO,I))/NJ
            DFNH3B=(NH3(LBRO,I+1)-NH3(LBRO,I))/NJ
            DFNH3K=(NH3(K,I+1)-NH3(K,I))/NJ
            NO3T=NO3(J,I)
            DFNO3S=(NO3(NR,I+1)-NO3(NR,I))/NJ
            DFNO3A=(NO3(LARO,I+1)-NO3(LARO,I))/NJ
            DFNO3B=(NO3(LBRO,I+1)-NO3(LBRO,I))/NJ
            DFNO3K=(NO3(K,I+1)-NO3(K,I))/NJ
            XNT=XN(J,I)
            DFXNS=(XNR(I+1)-XNR(I))/NJ
            DFXNA=(XN(LARO,I+1)-XN(LARO,I))/NJ
            DFXNB=(XN(LBRO,I+1)-XN(LBRO,I))/NJ
            DFXNK=(XN(K,I+1)-XN(K,I))/NJ
            DO 12 IJ=1,NJ,1
```

```
      SCVOT=SCVOD
      SCIST=KMST*SBST/(KSST+SBST)*FACT*XA(J,I)*DDT
      SYNTHT=SCVOT*P+SCIST
      DENCAP=SYNTHT/RNCOD
      DENIT=DENCAP
      RDEN=SCIST/SYNTHT
      ANPOTT=0.00
      IF((NO3T-DENIT).GT.0.00) GO TO 13
      DENIT=NO3T
      SYNTHT=DENIT*RNCOD
      SCIST=SYNTHT*RDEN
      SCVOT=SYNTHT*(1.0-RDEN)/P
      ANPOTT=DENCAP-DENIT
   13 CONTINUE
      SCVO=SCVO+SCVOT
      SCIS=SCIS+SCIST
      ANPOTS=ANPOTS+ANPOTT
      SBSS=SBS(NR,I)+(IJ-1)*DFSBSS
      SBSA=SBS(LARO,I)+(IJ-1)*DFSBSA
      SBSB=SBS(LBRO,I)+(IJ-1)*DFSBSB
      SBSK=SBS(K,I)+(IJ-1)*DFSBSK
      SBST=SBST+DUMIF*FCA*SBI(I)-SCIST+FCS*SCER
     1+DUMS*SBSS+DUMA*SBSA-DUMO*SBST
     1+DUMB*SBSB+DUMIR*SBSK
      IF(SBST.LT.0.00) SBST=0.00
      NITT=0.00
      NNIT=NNIT+NITT
      NH3S=NH3(NR,I)+(IJ-1)*DFNH3S
      NH3A=NH3(LARO,I)+(IJ-1)*DFNH3A
      NH3B=NH3(LBRO,I)+(IJ-1)*DFNH3B
      NH3K=NH3(K,I)+(IJ-1)*DFNH3K
      NO3S=NO3(NR,I)+(IJ-1)*DFNO3S
      NO3A=NO3(LARO,I)+(IJ-1)*DFNO3A
      NO3B=NO3(LBRO,I)+(IJ-1)*DFNO3B
      NO3K=NO3(K,I)+(IJ-1)*DFNO3K
      TNH3T=NH3T+DUMIF*NH3I(I)+CVNON-NITT
     1+DUMS*NH3S+DUMA*NH3A-DUMO*NH3T
     1+DUMB*NH3B+DUMIR*NH3K
     1+(1.0-FOE)*SCERN-(1.0-FOS)*FN*YH*SYNTHT
      IF(TNH3T.GT.0.0) GO TO 43
      NH3T=0.0
      NO3T=NO3T+DUMIF*NO3I(I)+NITT-DENIT+TNH3T
     1+DUMS*NO3S+DUMA*NO3A-DUMO*NO3T
     1+DUMB*NO3B+DUMIR*NO3K
      IF(NO3T.LT.0.00) NO3T=0.00
      GO TO 44
   43 NH3T=TNH3T
      NO3T=NO3T+DUMIF*NO3I(I)+NITT-DENIT
     1+DUMS*NO3S+DUMA*NO3A-DUMO*NO3T
     1+DUMB*NO3B+DUMIR*NO3K
      IF(NO3T.LT.0.00) NO3T=0.00
   44 XNS=XNR(I)+(IJ-1)*DFXNS
      XNA=XN(LARO,I)+(IJ-1)*DFXNA
      XNB=XN(LBRO,I)+(IJ-1)*DFXNB
      XNK=XN(K,I)+(IJ-1)*DFXNK
      XNT=XNT+YN*NITT-BNT*XNT*DDT
     1+DUMS*XNS+DUMA*XNA-DUMO*XNT
     1+DUMB*XNB+DUMIR*XNK
   12 CONTINUE
      SYNTH(J,I)=SCVO*P+SCIS
      ANPOT(J,I)=ANPOTS
      NIT(J,I)=NNIT
      SBS(J,I+1)=SBST
      NH3(J,I+1)=NH3T
```

```
128            NO3(J,I+1)=NO3T
129            XN(J,I+1)=XNT
130            DUMIF=DUMIF/RDT
131            DUMIR=DUMIR/RDT
132            DUMA=DUMA/RDT
133            DUMB=DUMB/RDT
134            DUMS=DUMS/RDT
135            DUMO=DUMO/RDT
136            BRAK=FR-(XS(J,I)/(FACT*XA(J,I)))
137            IF(BRAK.GT.0.00) GO TO 40
138            BRAK=0.00
139            XS(J,I)=FR*(FACT*XA(J,I))
140         40 XV(J,I)=XS(J,I)+XA(J,I)+XE(J,I)+XI(J,I)
141            SCIP=DT*KVT*BRAK*SBP(J,I)*FACT*XA(J,I)
142            SBP(J,I+1)=SBP(J,I)+DUMIF*(1.0-FCA)*SBI(I)-SCIP
143           1+DUMS*SBPR(I)+DUMA*SBP(LARO,I)-DUMO*SBP(J,I)
144           1+DUMB*SBP(LBRO,I)+DUMIR*SBP(K,I)
145           2+(1.0-FCS)*SCER/RDT
146            IF(SBP(J,I+1).LT.0.0) SBP(J,I+1)=0.0
147            XS(J,I+1)=XS(J,I)+SCIP/P-SCVO
148           1+DUMS*XSR(I)+DUMA*XS(LARO,I)-DUMO*XS(J,I)
149           1+DUMB*XS(LBRO,I)+DUMIR*XS(K,I)
150            IF(XS(J,I+1).LT.0.0) XS(J,I+1)=0.0
151            SU(J,I+1)=SU(J,I)+DUMIF*SUI(I)
152           1+DUMS*SU(NR,1)+DUMA*SU(LARO,I)-DUMO*SU(J,I)
153           1+DUMB*SU(LBRO,I)+DUMIR*SU(K,I)
154            NU(J,I+1)=NU(J,I)+DUMIF*NUI(I)
155           1+DUMS*NU(NR,I)+DUMA*NU(LARO,I)-DUMO*NU(J,I)
156           1+DUMB*NU(LBRO,I)+DUMIR*NU(K,I)
157            XA(J,I+1)=XA(J,I)+YH*SYNTH(J,I)-BHT*XA(J,I)*DT
158           1+DUMS*XAR(I)+DUMA*XA(LARO,I)-DUMO*XA(J,I)
159           1+DUMB*XA(LBRO,I)+DUMIR*XA(K,I)
160            XE(J,I+1)=XE(J,I)+F*BHT*XA(J,I)*DT
161           1+DUMS*XER(I)+DUMA*XE(LARO,I)-DUMO*XE(J,I)
162           1+DUMB*XE(LBRO,I)+DUMIR*XE(K,I)
163            XI(J,I+1)=XI(J,I)+DUMIF*XII(I)
164           1+DUMS*XIR(I)+DUMA*XI(LARO,I)-DUMO*XI(J,I)
165           1+DUMB*XI(LBRO,I)+DUMIR*XI(K,I)
166            NON(J,I+1)=NON(J,I)+DUMIF*NONI(I)-CVNON/RDT
167           1+DUMS*NON(NR,I)+DUMA*NON(LARO,I)-DUMO*NON(J,I)
168           1+DUMB*NON(LBRO,I)+DUMIR*NON(K,I)
169           2+FOE*SCERN/RDT-FOS*FN*YH*SYNTH(J,I)
170           3-FNS*(SCIP/P-SCVO)
171            AMON(J,I)=(CVNON/RDT+(1.0-FOE)*SCERN/RDT)
172           1-(1.0-FOS)*FN*YH*SYNTH(J,I)
173            ALK(J,I+1)=ALK(J,I)+DUMIF*ALKI(I)
174           1+DUMS*ALK(NR,I)+DUMA*ALK(LARO,I)-DUMO*ALK(J,I)
175           1+DUMB*ALK(LBRO,I)+DUMIR*ALK(K,I)
176           1+3.57*(SYNTH(J,I)/RNCOD+AMON(J,I))
177            XV(J,I+1)=XS(J,I+1)+XA(J,I+1)+XE(J,I+1)+XI(J,I+1)
178         11 CONTINUE
179            RETURN
180            END
```

```
      C*****************************************************************
            SUBROUTINE SETLCY(NL,J,QSR)
      C*****************************************************************
      C            THIS SUBROUTINE CALCULATES THE DYNAMIC RESPONSE OF THE
      C             SETTLING TANK
      C*****************************************************************
            PARAMETER NE=6
            PARAMETER NEE=7
            PARAMETER ND=241
            PARAMETER NDD=242
            COMMON/EXTC/SBS(NE,ND),SU(NE,ND),XA(NE,ND),XE(NE,ND),XI(NE,ND),
           1XV(NE,ND),XS(NE,ND),NU(NE,ND),NON(NE,ND),XN(NE,ND),NH3(NE,ND),
           2NO3(NE,ND),SBP(NE,ND),ALK(NE,ND)
            COMMON/BLOCKC/Q(NEE,NDD),V(NE),SBI(ND),SUI(ND),NIT(NE,ND),
           1NUI(ND),XII(ND),NONI(ND),NH3I(ND),NO3I(ND),SBPR(ND),ALKI(ND),
           2XAR(ND),XSR(ND),XER(ND),XIR(ND),XNR(ND),SYNTH(NE,ND),QAV(NEE),
           3ANPOT(NE,ND),AMON(NE,ND),DSFCTR(NE)
            REAL NUI,NONI,NH3I,NO3I,NIT
            REAL NU,NON,NH3,NO3
            DO 13 I=1,NL,1
            DUM=(QSR+Q(J+1,I))/QSR
            XAR(I)=XA(J,I)*DUM
            XSR(I)=XS(J,I)*DUM
            XER(I)=XE(J,I)*DUM
            XIR(I)=XI(J,I)*DUM
            SBPR(I)=SBP(J,I)*DUM
            XNR(I)=XN(J,I)*DUM
         13 CONTINUE
            RETURN
            END
```

```
      C*****************************************************************
            SUBROUTINE AROBCY(D,DT,J,FR,FCA,FCS,KVT,KMPT,KSPT,KMST,KSST,
           1P,NR,YH,BHT,F,KRT,FOE,FN,FOS,FRS,NJ,DDT,UMT,YN,KNT,RDT,BNT,
           2FACT,LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR)
      C*****************************************************************
      C              THIS SUBROUTINE CALCULATES THE DYNAMIC RESPONSE OF THE
      C                FIRST REACTOR IN A SERIES UNDER AEROBIC CONDITIONS
      C*****************************************************************
            PARAMETER NE=6
            PARAMETER NEE=7
            PARAMETER ND=241
            PARAMETER NDD=242
            INTEGER D
            DOUBLE PRECISION DT,DDT,RDT
            COMMON/EXTC/SBS(NE,ND),SU(NE,ND),XA(NE,ND),XE(NE,ND),XI(NE,ND),
           1XV(NE,ND),XS(NE,ND),NU(NE,ND),NON(NE,ND),XN(NE,ND),NH3(NE,ND),
           2NO3(NE,ND),SBP(NE,ND),ALK(NE,ND)
            COMMON/BLOCKC/Q(NEE,NDD),V(NE),SBI(ND),SUI(ND),NIT(NE,ND),
           1NUI(ND),XII(ND),NONI(ND),NH3I(ND),NO3I(ND),SBPR(ND),ALKI(ND),
           2XAR(ND),XSR(ND),XER(ND),XIR(ND),XNR(ND),SYNTH(NE,ND),QAV(NEE),
           3ANPOT(NE,ND),AMON(NE,ND),DSFCTR(NE)
            REAL KVT,KMPT,KSPT,KMST,KSST,KRT,KNT
            REAL NUI,NONI,NH3I,NO3I,NIT
            REAL NU,NON,NH3,NO3
            REAL NH3A,NH3S,NH3T,NH3B,NH3K,NITT,NNIT,NO3A,NO3S,NO3T,NO3B,NO3K
            DO 11 I=1,D,1
      C*****************************************************************
            CALL DUMVCY(J,I,DUMIF,DUMS,DUMA,DUMB,DUM1R,DUMO,
           1LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,
           1DT,DDT,RDT)
      C*****************************************************************
            K=J-1
            IF(K.EQ.0) K=1
            SCER=P*(1.0-F)*BHT*XA(J,I)*DDT
            SCERN=FN*SCER/P
            CVNON=KRT*XA(J,I)*NON(J,I)*DDT
            SCVOD=KMPT*XS(J,I)/(KSPT*XA(J,I)+XS(J,I)*P)*XA(J,I)*DDT
            SCVO=0.00
            SCIS=0.00
            NNIT=0.00
            SBST=SBS(J,I)
            DFSBSS=(SBS(NR,I+1)-SBS(NR,I))/NJ
            DFSBSA=(SBS(LARO,I+1)-SBS(LARO,I))/NJ
            DFSBSB=(SBS(LBRO,I+1)-SBS(LBRO,I))/NJ
            DFSBSK=(SBS(K,I+1)-SBS(K,I))/NJ
            NH3T=NH3(J,I)
            DFNH3S=(NH3(NR,I+1)-NH3(NR,I))/NJ
            DFNH3A=(NH3(LARO,I+1)-NH3(LARO,I))/NJ
            DFNH3B=(NH3(LBRO,I+1)-NH3(LBRO,I))/NJ
            DFNH3K=(NH3(K,I+1)-NH3(K,I))/NJ
            NO3T=NO3(J,I)
            DFNO3S=(NO3(NR,I+1)-NO3(NR,I))/NJ
            DFNO3A=(NO3(LARO,I+1)-NO3(LARO,I))/NJ
            DFNO3B=(NO3(LBRO,I+1)-NO3(LBRO,I))/NJ
            DFNO3K=(NO3(K,I+1)-NO3(K,I))/NJ
            XNT=XN(J,I)
            DFXNS=(XNR(I+1)-XNR(I))/NJ
            DFXNA=(XN(LARO,I+1)-XN(LARO,I))/NJ
            DFXNB=(XN(LBRO,I+1)-XN(LBRO,I))/NJ
            DFXNK=(XN(K,I+1)-XN(K,I))/NJ
            DO 12 IJ=1,NJ,1
            SCVOT=SCVOD
            SCIST=KMST*SBST/(KSST+SBST)*XA(J,I)*DDT
```

```
      SBSS=SBS(NR,I)+(IJ-1)*DFSBSS
      SBSA=SBS(LARO,I)+(IJ-1)*DFSBSA
      SBSB=SBS(LBRO,I)+(IJ-1)*DFSBSB
      SBSK=SBS(K,I)+(IJ-1)*DFSBSK
      SBST=SBST+DUMIF*FCA*SBI(I)-SCIST+FCS*SCER
     1+DUMS*SBSS+DUMA*SBSA-DUMO*SBST
     1+DUMB*SBSB+DUMIR*SBSK
      IF(SBST.LT.0.00) SBST=0.00
      SCVO=SCVO+SCVOT
      SCIS=SCIS+SCIST
      SYNTHT=SCVOT*P+SCIST
      NITT=(UMT/YN)*NH3T/(KNT+NH3T)*XNT*DDT
      NNIT=NNIT+NITT
      NH3S=NH3(NR,I)+(IJ-1)*DFNH3S
      NH3A=NH3(LARO,I)+(IJ-1)*DFNH3A
      NH3B=NH3(LBRO,I)+(IJ-1)*DFNH3B
      NH3K=NH3(K,I)+(IJ-1)*DFNH3K
      NO3S=NO3(NR,I)+(IJ-1)*DFNO3S
      NO3A=NO3(LARO,I)+(IJ-1)*DFNO3A
      NO3B=NO3(LBRO,I)+(IJ-1)*DFNO3B
      NO3K=NO3(K,I)+(IJ-1)*DFNO3K
      TNH3T=NH3T+DUMIF*NH3I(I)+CVNON-NITT
     1+DUMS*NH3S+DUMA*NH3A-DUMO*NH3T
     1+DUMB*NH3B+DUMIR*NH3K
     1+(1.0-FOE)*SCERN-(1.0-FOS)*FN*YH*SYNTHT
      IF(TNH3T.GT.0.0) GO TO 43
      NH3T=0.0
      NO3T=NO3T+DUMIF*NO3I(I)+NITT+TNH3T
     1+DUMS*NO3S+DUMA*NO3A-DUMO*NO3T
     1+DUMB*NO3B+DUMIR*NO3K
      GO TO 44
   43 NH3T=TNH3T
      NO3T=NO3T+DUMIF*NO3I(I)+NITT
     1+DUMS*NO3S+DUMA*NO3A-DUMO*NO3T
     1+DUMB*NO3B+DUMIR*NO3K
   44 XNS=XNR(I)+(IJ-1)*DFXNS
      XNA=XN(LARO,I)+(IJ-1)*DFXNA
      XNB=XN(LBRO,I)+(IJ-1)*DFXNB
      XNK=XN(K,I)+(IJ-1)*DFXNK
      XNT=XNT+YN*NITT-BNT*XNT*DDT
     1+DUMS*XNS+DUMA*XNA-DUMO*XNT
     1+DUMB*XNB+DUMIR*XNK
   12 CONTINUE
      SYNTH(J,I)=SCVO*P+SCIS
      NIT(J,I)=NNIT
      ANPOT(J,I)=0.00
      SBS(J,I+1)=SBST
      NH3(J,I+1)=NH3T
      NO3(J,I+1)=NO3T
      XN(J,I+1)=XNT
      DUMIF=DUMIF/RDT
      DUMIR=DUMIR/RDT
      DUMA=DUMA/RDT
      DUMB=DUMB/RDT
      DUMS=DUMS/RDT
      DUMO=DUMO/RDT
      BRAK=FR-(XS(J,I)/(XA(J,I)))
      IF(BRAK.GT.0.00) GO TO 40
      BRAK=0.00
      XS(J,I)=FR*(XA(J,I))
   40 XV(J,I)=XS(J,I)+XA(J,I)+XE(J,I)+XI(J,I)
      SCIP=DT*KVT*BRAK*SBP(J,I)*XA(J,I)
      SBP(J,I+1)=SBP(J,I)+DUMIF*(1.0-FCA)*SBI(I)-SCIP
     1+DUMS*SBPR(I)+DUMA*SBP(LARO,I)-DUMO*SBP(J,I)
```

```
     1+DUMB*SBP(LBRO,I)+DUMIR*SBP(K,I)
     2+(1.0-FCS)*SCEP/RDT
      IF(SBP(J,I+1).LT.0.0) SBP(J,I+1)=0.0
      XS(J,I+1)=XS(J,I)+SCIP/P-SCVO
     1+DUMS*XSR(I)+DUMA*XS(LARO,I)-DUMO*XS(J,I)
     1+DUMB*XS(LBRO,I)+DUMIR*XS(K,I)
      IF(XS(J,I+1).LT.0.0) XS(J,I+1)=0.0
      SU(J,I+1)=SU(J,I)+DUMIF*SUI(I)
     1+DUMS*SU(NR,I)+DUMA*SU(LARO,I)-DUMO*SU(J,I)
     1+DUMB*SU(LBRO,I)+DUMIR*SU(K,I)
      NU(J,I+1)=NU(J,I)+DUMIF*NUI(I)
     1+DUMS*NU(NR,I)+DUMA*NU(LARO,I)-DUMO*NU(J,I)
     1+DUMB*NU(LBRO,I)+DUMIR*NU(K,I)
      XA(J,I+1)=XA(J,I)+YH*SYNTH(J,I)-BHT*XA(J,I)*DT
     1+DUMS*XAR(I)+DUMA*XA(LARO,I)-DUMO*XA(J,I)
     1+DUMB*XA(LBRO,I)+DUMIR*XA(K,I)
      XE(J,I+1)=XE(J,I)+F*BHT*XA(J,I)*DT
     1+DUMS*XER(I)+DUMA*XE(LARO,I)-DUMO*XE(J,I)
     1+DUMB*XE(LBRO,I)+DUMIR*XE(K,I)
      XI(J,I+1)=XI(J,I)+DUMIF*XII(I)
     1+DUMS*XIR(I)+DUMA*XI(LARO,I)-DUMO*XI(J,I)
     1+DUMB*XI(LBRO,I)+DUMIR*XI(K,I)
      NON(J,I+1)=NON(J,I)+DUMIF*NONI(I)-CVNON/RDT
     1+DUMS*NON(NR,I)+DUMA*NON(LARO,I)-DUMO*NON(J,I)
     1+DUMB*NON(LBRO,I)+DUMIR*NON(K,I)
     2+FOE*SCERN/RDT-FOS*PN*YH*SYNTH(J,I)
     3-PNS*(SCIP/P-SCVO)
      AMON(J,I)=(CVNON/RDT+(1.0-FOE)*SCERN/RDT)
     1-(1.0-FOS)*PN*YH*SYNTH(J,I)
      ALK(J,I+1)=ALK(J,I)+DUMIF*ALKI(I)
     1+DUMS*ALK(NR,I)+DUMA*ALK(LARO,I)-DUMO*ALK(J,I)
     1+DUMB*ALK(LBRO,I)+DUMIR*ALK(K,I)
     1-7.14*NIT(J,I)+3.57*AMON(J,I)
      XV(J,I+1)=XS(J,I+1)+XA(J,I+1)+XE(J,I+1)+XI(J,I+1)
   11 CONTINUE
      RETURN
      END
```

APPENDIX A2

COMPUTER PROGRAM FOR THE UNSTEADY STATE

```
C*****************************************************************
C
C              UNSTEADY STATE PROGRAM AS AT 5-5-1981
C
C*****************************************************************
C*****************************************************************
C
C          ''THIS PROGRAMME IS THE JOINT PROPERTY OF THE
C          UNIVERSITY OF CAPE TOWN AND THE WATER RESEARCH
C          COMMISSION OF SOUTH AFRICA. BEFORE USER MAY
C          LEGALLY UTILIZE IT, HE MUST OBTAIN THE NECESSARY
C          AUTHORIZATION, I.E.:
C
C          (A) IN THE CASE OF STUDENTS AT THE UNIVERSITY OF
C              CAPE TOWN WHO WILL USE THE PROGRAMME FOR
C              EDUCATIONAL PURPOSES ONLY, THIS AUTHORIZATION
C              CAN BE OBTAINED FROM PROF. G. VAN R. MARAIS
C              OF THE UNIVERSITY OF CAPE TOWN, AND
C          (B) IN ALL OTHER CASES, WRITTEN AUTHORIZATION
C              MUST BE OBTAINED FROM BOTH THE UNIVERSITY
C              OF CAPE TOWN (PRIVATE BAG, RONDEBOSCH, 7700,
C              CAPE, SOUTH AFRICA) AND THE WATER RESEARCH
C              COMMISSION (P.O. BOX 824, PRETORIA, 0001,
C              SOUTH AFRICA)''
C
C*****************************************************************
C*****************************************************************
C          THIS PROGRAM IS USED TO SIMULATE THE RESPONSE OF SERIES
C          REACTOR ACTIVATED SLUDGE SYSTEMS UNDER UNSTEADY STATE
C          CONDITIONS.THE REACTORS ARE ASSUMED TO BE COMPLETELY
C          MIXED . THE VALUES OF THE VARIABLES ARE CALCULATED AS
C          A FUNCTION OF TIME USING THE SUPPLIED STARTING VALUE
C          AND THE KINETIC EXPRESSIONS DEVELOPED BY DOLD, EKAMA
C          AND MARAIS FOR AN AEROBIC ENVIRONMENT (I.E. THE BI-
C          SUBSTRATE /DEATH REGENERATION/ACTIVE SITE THEORY) OR
C          THOSE LISTED IN TABLE (4.1 CHAPTER 4) FOR AN ANOXIC
C          ENVIRONMENT. THE CALCULATED VALUES OF ALL THE VARIABLES
C          IN EACH REACTOR CAN BE PRINTED AS A FUNCTION OF TIME
C          AND/OR WRITTEN TO A PLOT FILE FOR SUBSEQUENT PLOTTING
C          TOGETHER WITH EXPERIMENTAL DATA.THE PLOTTING PROGRAM IS
C          LISTED IN APPENDIC A.6
C          THE RESPONSE IS CALCULATED OVER THE PERIOD THAT INPUT
C          FLOW AND LOAD DATA IS SUPPLIED
C          THE REQUIRED PERIOD OF PLOTTED RESPONSE MUST BE SPECIFIED
C          THE MAIN PROGRAM HAS FOUR SUBROUTINES:
C        1)UDUMV CALCULATES THE HYDRAULIC EFFCTS IN THE REACTORS
C        2)UAERO CALCULATES THE RESPONSE OF A REACTOR IN AN AEROBIC
C          ENVIRONMENT
C        3)UANOX CALCULATES THE RESPONSE OF A REACTOR IN AN ANOXIC
C          ENVIRONMENT
C        4)USETL CALCUALTES THE RESPONSE OF THE SETTLER
C
C           THE PROGRAM AS LISTED BELOW IS SUBJECTED TO THE FOLLOWING
C           LIMITATIONS
C           1)THE LENGTHS OF THE ANOXIC AND THE AEROBIC PERIODS IN
C             A REACTOR ARE FOR ONE CYCLE CAN EACH ASSUME ONE VALUE ONLY
C           2)THE INFLUENT FOLW CAN ONLY BE INTRODUCED INTO ONE
C             REACTOR
C           3) THE RECYCLE FLOW FROM THE SETTLER CAN ONLY BE CONVEYED
C              TO ONE REACTOR (WHICH IS NOT NECESSARILY THE FIRST ONE)
C           4) IT IS ONLY POSSIBLE TO HAVE TWO INTERNAL RECYCLES FLOWS
C             (A AND B) EACH OF WHICH CONVEYS MIXED LIQUOR FROM ONE
C             REACTOR TO ANOTHER THE LATTER BEING NEARER TO THE REACTOR
```

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      C              RECIEVING THE INFLUENT
      C           5) ALL RECYCLE FLOWS ARE CONSTANT WITH TIME.
      C           6) OPTIONS FOR THE INFLUENT FLOW PATTERN ARE:
      C              A)SINOIDAL
      C              B)SQUARE WAVE
      C              C)GENERAL PATTERN WITH LINEAR INTERPOLATION
      C           7)THE CONCENTRATION PATTERNS OF INFLUENT COD,TKN,NITRATE
      C             AND ALKALINITY HAVE THE SAME LIMITATIONS AS MENTIONED
      C             FOR THE INFLUENT FLOW UNDER 6
      C           8)IT IS ASSUMED THAT DURING A CYCLE THE CONSTANTS CHARA
      C             CHARACTERIZING THE SEWAGE (FUS,FUP,FCA,FUN,FAN,SN) DO
      C             NOT CHANGE
      C           9)IT IS ASSUMED THAT THE D.O. CONCENTRATION IN THE AEROBIC
      C             REACTORS IS SUFFICIENT SO THAT IT DOES NOT LIMIT THE
      C             RATES OF UTILIZATION OF ORGANIC SUBSTRATE OR OF NITRI-
      C             FICATION
      C          10)IT IS ASSUMED THAT ANY D.O. RECYCLED TO YHE ANOXIC
      C             REACTORS DOES NOT AFFECT THE RATE OF DENITRIFICATION
      C          11)THE PH AND TEMPERATURE ARE ASSUMED TO BE CONSTANT
      C             OVER ONE CYCLE AND THE SAME IN ALL REACTORS
      C          12)IS IS ASSUMED THAT THE PROCESS IS A SINGLE SLUDGE
      C             PROCESS I.E. THERE IS ONLY ONE SETTLER
      C          13) IT IS ASSUMED THAT SLUDGE WASTAGE TAKES PLACE FROM
      C              ONE OR MORE REACTORS.THE FLOW OF WASTED SLUDGE IS CON-
      C             STANT DURING THE PERIOD OF WASTAGE AND PROPORTIONAL
      C             TO THE REACTOR VOLUME.
      C           14) THE SETTLER IS TAKEN AS AN IDEAL LIQUID SOLID
      C              SEPARATOR, I.E. THE RETENTION TIME IN THE SETTLER
      C              IS ZERO AND NO REACTION TAKES PLACE IN THE SETTLER.
      C
      C             ALL THE LIMITATIONS ABOVE ,EXCEPT THE LAST ONE, ARE
      C            IMPOSED TO REDUCE THE REQIRED CORE STORAGE. IF
      C            REQUIRED THR PROGRAM CAN BE REWRITTEN IN SUCH
      C            FASHION THAT NONE OF THE LIMITATIONS 1 TO 13 APPLY.
      C            THE FOURTEENTH LIMITATION CAN BE OMITTED ONLY IF SUFF
      C            SUFFICIENT DATA IS AVAILABLE TO DESCRIBE THE
      C            SETTLING BEHAVOUR UNDER CYCLIC FLOW AND LOAD
      C            CONDITIONS.
      C
      C          NOTE THAT THIS PROGRAM CAN ONLY BE USED TO CHECK A
      C          DESIGN AND HENCE REQUIRES A DESIGN CONFIGURATION
      C          WITH SPECIFIED REACTOR VOLUMES.
      C
      C          THE PROGRAM HAS BEEN TESTED FOR DIFFERENT VARIANTS
      C          OF THE ACTIVATED SLUDGE PROCESS, I.E.
      C          1)SERIES SUSPENSION MIXED AERATED LAGOONS
      C          2)CONTACT STABILIZATION PROCESS
      C          3)OXIDATION DITCH
      C          4)SERIES COMPLETELY NIXED REACTOR SYSTEM(1 TO 6 REACTORS)
      C          WITH AND WITHOUT ANOXIC REACTORS.
      C
      C*************************************************************************
      C*************************************************************************
      C
      C          INPUT OF DATA
      C          -------------
      C
      C          FIRST CARD:
      C          -----------
      C
      C          TYPE = LOGICAL NUMBER DESCRIBING INFLUENT LOAD PATTERN
      C            IF = 1    SINE WAVE LOAD PATTERN
      C            IF = 2    SQUARE WAVE LOAD PATTERN
      C            IF = 3    GENERAL LOADING PATTERN
```

```
C        RESULT = LOGICAL NUMBER TO DIRECT OUTPUT
C            IF = 1     PRINT RESULTS
C            IF = 2     PLOT RESULTS
C            IF = 0     PLOT AND PRINT RESULTS
C           KOM = TIME AT WHICH TIME BASE COUNTER COMMENCES
C           NDH = LENGTH OF PERIOD OF PLOTTED SIMULATION
C             M = LENGTH OF PERIOD OF CALCULATED SIMULATION
C             DATE,XMONTH AND YEAR REFER TO THE DATE THE EXPERIMENT
C             TO BE SIMULATED WAS CARRIED OUT.
C
C
C             SECOND CARD
C             -----------
C             (INPUT FLOW AND LOAD)
C            Q0 = AVERAGE INFLUENT FLOW FOR SINE WAVE PATTERN
C          STI0 = INFLUENT COD CONCENTRATION DURING PEAK FLOW PERIOD FOR
C                 SQUARE WAVE PATTERN OR AVERAGE FOR SINE WAVE PATTERN
C          NTI0 = INFLUENT TKN CONCENTRATION DURING PEAK FLOW PERIOD FOR
C                 SQUARE WAVE PATTERN OR AVERAGE FOR SINE WAVE PATTERN
C         NO3I0 = INFLUENT NO3 CONCENTRATION DURING PEAK FLOW PERIOD FOR
C                 SQUARE WAVE PATTERN OR AVERAGE FOR SINE WAVE PATTERN
C         ALKI0 = INFLUENT ALK CONCENTRATION DURING PEAK FLOW PERIOD FOR
C                 SQUARE WAVE PATTERN OR AVERAGE FOR SINE WAVE PATTERN
C        VARSTI = AMPLITUDE OF INPUT COD CONC. WAVE (SINE WAVE)
C        VARNTI = AMPLITUDE OF INPUT TKN CONC. WAVE (SINE WAVE)
C         VNO3I = AMPLITUDE OF INPUT NO3 CONC. WAVE (SINE WAVE)
C          VARQ = AMPLITUDE OF INFLUENT FLOW WAVE (SINE WAVE)
C         STI00 = INFLUENT COD CONCENTRATION DURING BASE FLOW PERIOD
C         NTI00 = INFLUENT TKN CONCENTRATION DURING BASE FLOW PERIOD
C        NO3I00 = INFLUENT NO3 CONCENTRATION DURING BASE FLOW PERIOD
C        ALKI00 = INFLUENT ALK CONCENTRATION DURING BASE FLOW PERIOD
C           LFP = LENGTH OF FEED PERIOD (SQUARE WAVE)
C            PQ = PEAK FLOW RATE IN SQUARE WAVE PATTERN
C            BQ = BASE FLOW RATE IN SQUARE WAVE PATTERN
C            VF = VOLUME OF FEED PER DAY
C         START = BEGIN FEEDING PERIOD (SQ.WAVE)
C
C           THIRD CARD
C           ----------
C           (KINETIC CONSTANTS AND MASS PARAMETERS)
C
C            AN EXTRA T OR 20 INDICATES VALUES AT T OR 20 DEG. C
C             KV = MAX. RATE OF COD TRANSFER FROM SEWAGE INTO STORAGE
C             KR = RATE OF CONVERSION OF ORGANIC N TO SALINE NH3
C            KMP = MAX. RATE OF  ORGANISM SYNTHESIS FROM STORED COD
C            KMS = MAX. RATE OF  ORGANISM SYNTHESIS FROM SOLUBLE COD
C            KSP = HALF SATURATION COEF. FOR STORED COD  UTILIZATION
C            KSS = HALF SATURATION COEF. FOR SOLUBLE COD  UTILIZATION
C             UM = MAXIMUM GROWTH RATE OF NITRIFYING ORGANISMS
C             KN = HALF SATURATION COEF. FOR NH3  UTILIZATION
C             BH = DEATH RATE
C             BN = DEATH RATE FOR NITRIFICATION
C             YH = YIELD OF ORGANISMS FROM COD UTILIZED
C             YN = ACTIVE MASS YIELD OF NITRIFYING ORGANISMS FROM AMMONIA
C              F = UNBIODEG. FRACTION OF ORGANISM
C             FR = MAXIMUM FRACTION OF STORED COD (AS VSS) TO ACTIVE
C                  ORGANISM MASS
C             FN = FRACTION OF CARBONACEOUS CELL MASS AS N
C            FNS = FRACTION OF NITROGEN ASSOCIATED WITH STORED COD
C              P = COD EQUIVALENT OF VSS
C            FCA = RATIO OF EASILY (SOLUBLE) TO TOTAL INF. BIODEG. MAT.
C            FOE = FRACTION OF N RELEASED BY DEATH
C                  AS ORGANIC NITROGEN
C            FOS = FRACTION OF NITROGEN REQUIRED FOR CELL SYNTHESIS
```

```
C              AS ORGANIC NITROGEN
C          FACT = RATIO OF KMP IN AN ANOXIC AND IN AN AEROBIC ENV.
C           FCS = FRACTION OF LYSED BIODEG. MATERIAL RELEASED AS
C              EASILE BIODEGRADABLE MATERIAL
C
C
C           FOURTH CARD
C           -----------
C           (OPERATIONAL PARAMETERS AND SEWAGE CHARACTERISTICS)
C              RS = SLUDGE AGE
C               NR= NUBER OF REACTORS IN SERIES
C              D  = NUMER OF INTEGRATION STEPS PER DAY
C              NJ = NUMBER OF SUBSTEP FOR UTILIZARION OF SBS AND NH3
C             FUS = FRACTION OF SOLUBLE UNBIODEG. COD
C             FUP = FRACTION OF SOLID UNBIODEG. COD AS VSS
C              UN = UNBIODEGRADABLE FRACTION OF N
C            SN = FRACTION OF TKN AS AMMONIA
C
C           FIFTH CARD
C           ----------
C           (TEMPERATURE DEPENDENCIES , PH AND TEMP.)
C
C           THEA = ARRHENIUS TEMPERATURE DEP. COEF. FOR KMS
C           THEM = ARRHENIUS TEMPERATURE DEP. COEF. FOR KMP
C           THES = ARRHENIUS TEMPERATURE DEP. COEF. FOR KSP
C           THEE = ARRHENIUS TEMPERATURE DEP. COEF. FOR BHT
C           THEN = ARRHENIUS TEMPERATURE DEP. COEF. FOR UNMT
C           THEZ = ARRHENIUS TEMPERATURE DEP. COEF. FOR KSS
C          PHIN = PH DEPENDENCY OF NITRIFICATION RATE
C        TEMP = MIXED LIQUOR TEMPERATURE
C          PH = MIXED LIQUOR PH
C
C           SIXTH CARD
C           ----------
C           (REACTOR VOLUMES AND SLUDG WASTAGE)
C
C            V = VOLUME OF REACTOR
C         LRSD = LOGICAL NUMBER DESCRIBING SLUDGE WASTAGE
C              FROM REACTORS
C           IF = 1  SLUDGE TO BE WASTED FROM REACTOR
C           IF = 0  NO SLUDGE TO BE WASTED FROM REACTOR
C
C
C           SEVENTH CARD
C           ------------
C
C          ICSD = TIME AT WHICH SLUDGE WASTAGE COMMENCES
C          ISSD = TIME AT WHICH SLUDGE WASTAGE TERMINATES
C          NRSD = NUMBER OF REACTORS FROM WHICH SLUDGE IS WASTED
C        LINI = REACTOR NUMBER RECIEVING INFLUENT FLOW
C         LSRI = REACTOR NUMBER RECIEVING UNDERFLOW
C        LARI = REACTOR NUMBER RECIEVING A-RECYCLE
C        LBRI = REACTOR NUMBER RECIEVING B-RECYCLE
C        LARO = REACTOR NUMBER FROM WHICH LEAVES A-RECYCLE
C        LBRO = REACTOR NUMBER FROM WHICH LEAVES B-RECYCLE
C          SR = UNDERFLOW RECYCLE  RATIO
C          AR = A-RECYCLE RATIO
C          BR = B-RECYCLE RATIO
C
C           EIGHTH CARD
C           -----------
C           (ANOXIC SLUDGE MASS FRACTION)
C           INIT = INITIAL AEROBIC PERIOD
C           ANOX = DURATION OF AN ANOXIC PERIOD]
```

```
C          AROB = DUARTION OF AN AEROBIC PERIOD
C       NOTE
C       ----
C          NITROGEN CONCENTRATIONS MEASURED IN MG-N/L WITH TKN TEST
C          COD CONCENTRATIONS MEASURED IN MG/L OF COD
C          VSS CONCENTRATIONS MEASURED IN MG/L OF VSS
C          OXYGEN CONSUMPTIONS IN MG/L/HOUR
C          UNIT OF MASS IS THE MG
C          UNIT OF VOLUME IS THE L
C          UNIT OF TIME IS THE DAY
C          SYMBOL S REFERS TO COD
C          SYMBOL X REFERS TO VSS
C          SYMBOL N REFERS TO NITROGEN
C          SYMBOL O REFERS TO OXYGEN
C
C
C
C       DIMENSION STATEMENTS
C       --------------------
C
      PARAMETER NE=3
      PARAMETER NEE=4
      PARAMETER ND=241
      PARAMETER NDD=242
      COMMON/EXTC/SBS(NE,ND),SU(NE,ND),XA(NE,ND),XE(NE,ND),XI(NE,ND),
     1XV(NE,ND),XS(NE,ND),NU(NE,ND),NON(NE,ND),XN(NE,ND),NH3(NE,ND),
     2NO3(NE,ND),SBP(NE,ND),ALK(NE,ND)
      COMMON/BLOCKC/Q(NEE,NDD),V(NE),SBI(ND),SUI(ND),NIT(NE,ND),
     1NUI(ND),XII(ND),NONI(ND),NH3I(ND),NO3I(ND),SBPR(ND),ALKI(ND),
     2XAR(ND),XSR(ND),XER(ND),XIR(ND),XNR(ND),SYNTH(NE,ND),QAV(NEE),
     3ANPOT(NE,ND),AMON(NE,ND)
      CHARACTER IMFT*19/'((20X,I4, (E13.5)))'/,CH*1
      REAL KVT,KMPT,KSPT,KMST,KSST,KRT,KNT,NTITI,NTITO
      REAL NUI,NONI,NH3I,NO3I,NIT
      REAL NU,NON,NH3,NO3
      INTEGER D,DD,HK,TYPE,RESULT,TN
      DOUBLE PRECISION DT,DDT,RDT
      DIMENSION STI(ND),STQ(ND)
      REAL NTI(ND),NTQ(ND),NO3Q(ND),NXII(ND)
      REAL NTIAV,NUIAV,NONIAV,NO3IAV,NH3IAV,NXIIAV
      REAL NTQAV,NO3QAV
      REAL NTIMAX,NO3IAX,NTQMAX,NO3QAX
      REAL NTI0,NTI00,NO3I0,NO3I00,LFP
      DIMENSION RA(NE),RN(NE),LRSD(NE),NTDN(NE,ND),INTDN(NE),ALKQ(ND)
      REAL KV20,KMP20,KSP20,KMS20,KSS20,KNMAX,KN20,KR20
      DIMENSION KO(ND),WQ(NE,ND),INIT(NE),ANOX(NE),AROB(NE),FANOX(NE),
     1OC(NE,ND),ON(NE,ND),OT(NE,ND),ST(NE,ND),RXS(NE,ND)
      REAL NT(NE,ND)
      DIMENSION SBSAV(NE),ODAV(NE),OCAV(NE),RNO3(NE),ALKAV(NE),AMAV(NE),
     1ONAV(NE),OTAV(NE),STAV(NE),SUAV(NE),SBPAV(NE),
     2XAAV(NE),XEAV(NE),XSAV(NE),XIAV(NE),XVAV(NE),
     3XNAV(NE),RXSAV(NE),NAME(NE),LNAME(3),APTAV(NE),APTNR(NE)
      REAL NONAV(NE),NTAV(NE),NO3AV(NE),NH3AV(NE),NUAV(NE),NSWAV(NE)
      REAL NONSS,NUSS,NO3SS,NH3SS
C
C
C          INPUT OF DATA
C          -------------
  100 FORMAT ( )
  101 FORMAT(2I1,I4,I3,I4,A2,A3,A4)
      READ 101,TYPE,RESULT,KOM,NDH,M,DATE,XMONTH,YEAR
      WRITE(5,101)TYPE,RESULT,KOM,M,DATE,XMONTH,YEAR
      IF(TYPE.EQ.1) GO TO 111
      IF(TYPE.EQ.2) GO TO 112
```

```
      IF(TYPE.EQ.3) GO TO 113
  111 READ 100,PI
      WRITE (5,100)PI
      READ 100,Q0,STI0,NTI0,NO3I0,ALKI0,VARQ,VARSTL,VARNTL,VNO3L,VALKI
      GO TO 113
  112 READ 100,VF,STI0,NTI0,NO3I0,ALKI0,BQ,STI00,NTI00,NO3I00,ALKI00,LFP
     &,START
  113 READ(8,100) KV20,KNP20,KSP20,KMS20,KSS20,UMAX,KEMAX,BH20,BN20,
     1KR20,FOS,FCA,FR,YH,YN,F,P,FH,FOR,FACT,FCS
      READ 100,RS,NR,D,NJ,FUS,FUP,UN,SN
      READ 100,THEA,THEM,THES,THEE,PHIN,THEN,THEZ,TEMP,PH
      READ(8,100) (V(J),LRSD(J),J=1,NR,1)
      READ(8,100) ICSD,ISSD,KRSD,LINI,LSRI,LARI,LARO,LBRI,LBRO,SK,AR,BR
      READ(8,100)(INIT(J),ANOX(J),AROB(J),J=1,NR)
      KH=D/24
      NI=D+1
      NII=D+2
      DD=NJ*D
      NRR=NR+1
C
C
C           CALCULATION CHARACTERISTICS
C           ---------------------------
C           THIS PROGRAMME REQUIRES THE FOLLOWING CORE STORAGE CAPACITY:-
C            FOR ND=241 AND NF=4*(ND-1)+1 AND NEE=NE+1
C            STORAGE=10*NEE+23 KILOWORDS
C            KO = TIME BASE VARIABLE
C             D = NUMBER OF INTERGRATION STEPS PER DAY
C            DT = INTERGRATION STEP LENGTH
C            DD = NUMBER OF INTERGRATION STEPS PER DAY FOR NITRIFICATION
C           DDT = INTERGRATION STEP LENGTH FOR NITRIFICATION
C           RDT = RATIO OF INTERGRATION STEP LENGTHS
C            KH = NUMBER OF TIME STEPS PER HOUR
C            HK = NO. OF INTERVALS PER DAY  OF SLUDGE WASTAGE
C
C           SEWAGE CHARACTERISTICS
C           ----------------------
C             Q = INFLUENT FLOW
C           STI = TOTAL INFLUENT COD
C           NTI = TOTAL INFLUENT TKN
C           STQ = TOTAL INFLUENT COD LOAD
C           NTQ = TOTAL INFLUENT TKN LOAD
C          NO3Q = TOTAL INFLUENT NO3 LOAD
C          NONI = INFLUENT ORGANIC NITROGEN CONCENTRATION
C          NH3I = INFLUENT AMMONIA CONCENTRATION
C          NO3I = INFLUENT NITRATE CONCENTRATION
C           NUI = UNBIODEGRADABLE FRACTION OF SEWAGE N
C          NXII = UNSABLE N IN INERT MATERIAL IN INFLUENT
C           XII = UNBIODEG. SOLID FRACTION OF SEWAGE COD AS VSS
C           SBI = BIODEG. FRACTION OF SEWAGE COD
C           SUI = UNBIODEG. SOLUBLE FRACTION OF SEWAGE COD
C         NM,NN = LIMITS OF PEAK FLOW PERIOD
C          QMAX = MAXIMUM FLOW VALUE OF THE DAY
C         STIMAX = MAXIMUM COD CONCENTRATION VALUE OF THE DAY
C        NTIMAX = MAXIMUM TKN CONCENTRATION VALUE OF THE DAY
C        NO3IAX = MAXIMUM NO3 CONCENTRATION VALUE OF THE DAY
C        STQMAX = MAXIMUM COD LOAD VALUE OF THE DAY
C        NTQMAX = MAXIMUM TKN LOAD VALUE OF THE DAY
C        NO3QAX = MAXIMUM NO3 LOAD VALUE OF THE DAY
C        VARSTL = AMPLITUDE OF INFLUENT COD LOAD WAVE (SINE WAVE)
C        VARNTL = AMPLITUDE OF INFLUENT TKN LOAD WAVE (SINE WAVE)
C        VRNO3L = AMPLITUDE OF INFLUENT NO3 LOAD WAVE (SINE WAVE)
C
C           PROCESS CHARACTERISTICS
```

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C          ------------------------
C
C       NTDN = LOGICAL NUMBER DESCRIBING PROCESS OPERATION
C          IF = 1  PROCESS INCLUDES NITRIFICATION ONLY
C          IF = 1 REACTOR IS IN AN AEROBIC ENVIRONMENT
C          IF = 0 REACTOR IS IN AN ANOXIC ENVIRONMENT
C        TEMP = OPERATING TEMPERATURE OF MIXED LIQUOR
C          PH = OPERATING PH OF MIXED LIQUOR
C         RSM = MINIMUM SLUDGE AGE FOR NITRIFICATION
C          RN = NOMINAL HYDRAULIC RETENTION TIME IN REACTORS
C          RA = ACTUAL HYDRAULIC RETENTION TIME IN REACTORS
C         TRN = TOTAL HYDRAULIC RETENTION TIME OF PROCESS
C         QSR = SLUDGE UNDERFLOW RECYCLE RATE
C         QAR = MIXED LIQUOR RECYCLE RATE (A-RECYCLE)
C         QBR = MIXED LIQUOR RECYCLE RATE (B-RECYCLE)
C          NR = NUMBER OF REACTORS IN SERIES
C         NRR = NUMBER OF REACTORS IN SERIES INCLUDING CLARIFIER
C          VP = TOTAL VOLUME OF PROCESS
C         END = END FEEDING PERIOD (SQ.WAVE)
C          WQ = SLUDGE WASTE FLOW
C     N.B.  THE SLUDGE WASTE FLOW MAY NEVER BE GREATER THAN
C           THE INFLUENT FLOW AT ANY INERVAL OF THE DAY
C       I.E. THE EFFLUENT FLOW FROM A REACTOR MAY NEVER BE LESS
C           THAN ZERO AT ANY INTERVAL OF THE DAY
C
C          GENERAL NOTATION FOR PROCESS VARIABLES IN REACTOR
C          -------------------------------------------------
C
C          SBP = BIODEGRADABLE PARTICULATE COD
C          SBS = BIODEGRADABLE SOLUBLE COD
C           SU = UNBIODEGRADABLE COD
C           ST = TOTAL COD
C           XS = STORED COD
C           XA = ACTIVE ORGANISMS
C           XE = INERT RESIDUE FROM DECAYING ORGANISMS
C           XI = INERT MATERIAL FROM SEWAGE
C           XN = ACTIVE NITROSOMONAS
C           XV = TOTAL VSS
C           OA = OXYGEN CONSUMPTION RATE FOR ADSORPTION OF COD
C           OS = OXYGEN CONSUMPTION RATE FOR SYNTHESIS
C           OE = OXYGEN CONSUMPTION RATE FOR ENDOGENOUS RESPIRATION
C           OC = TOTAL CARBONACEOUS OXYGEN CONSUMPTION RATE
C           ON = OXYGEN CONSUMPTION FOR NITRIFICATION
C           OT = TOTAL OXYGEN CONSUMPTION RATE
C         AMON = AMMONIFICATION RATE (MG N/L/D)
C          NSW = NITROGEN REMOVED IN SLUDGE WASTAGE PER DAY
C           NT = TOTAL TKN IN REACTOR
C           NU = UNBIODEGRADABLE TKN IN REACTOR
C          NON = ORGANIC NITROGEN CONCENTRATION  IN REACTOR
C          NH3 = AMMONIA CONCENTRATION IN REACTOR
C          NO3 = NITRATES CONCENTRATION IN REACTOR
C          ALK = ALKALINITY CONCENTRATION IN REACTOR
C          SCI = ENERGY ENTERING STORAGE FROM LIQUID PHASE (MG-COD/L)
C         SCVO = ENERGY REMOVED FROM STORAGE DUE TO CELL SYNTHESIS
C
C          THE SUFFIX 'AV' AFTER A SYMBOL INDICATES DAILY AVERAGE
C           VALUES FOR VARIABLES IN INFLUENT OR REACTORS
C          THE SUFFIX 'R' AFTER A SYMBOL INDICATES
C           VALUES FOR VARIABLES IN RECYCLE
C          THE SUFFIX 'P' AFTER A SYMBOL INDICATES VALUES
C           FOR VARIABLES CARRIED OVER TO PLOTTER ROUTINES
C          THE SUFFIX 'MAV' AFTER A SYMBOL INDICATES DAILY AVERAGE
C           VALUES OF VARIABLES FOR PROCESS
C
```

```
C          BALC = PERCENTAGE COD RECOVERY
C         TCODI = TOTAL COD MASS INPUT
C         TCODO = TOTAL COD MASS OUTPUT
C         TNTI = TOTAL TKN MASS INPUT
C         TNTO = TOTAL TKN MASS OUTPUT
C         TALKI = TOTAL ALKALINITY MASS INPUT
C         TALKO = TOTAL ALKALINITY MASS OUTPUT
C         TWCOD = TOTAL COD MASS IN WASTE FLOWS FROM REACTORS
C         TWN  = TOTAL TKN MASS IN WASTE FLOWS FROM REACTORS
C         TWALK = TOTAL ALK MASS IN WASTE FLOWS FROM REACTORS
C          WCOD = COD MASS IN WASTE FLOW FROM REACTOR
C          ECOD = TOTAL COD MASS IN EFFLUENT
C          BALN = PERCENTAGE NITROGEN RECOVERY
C          TNTI = TOTAL NITROGEN MASS INPUT
C          TNTO = TOTAL NITROGEN MASS OUTPUT
C           TWN = TOTAL NITROGEN MASS IN WASTE FLOWS FROM REACTORS
C            WN = NITROGEN MASS IN WASTE FLOW FROM REACTOR
C          ECOD = TOTAL COD IN EFFLUENT
C            EN = TOTAL NITROGEN MASS IN EFFLUENT
C          EALK = TOTAL ALKALINTY IN EFFLUENT
C
      IF(TYPE.EQ.1) GO TO 115
      IF(TYPE.EQ.2) GO TO 116
      READ(8,100) (Q(1,I),I=1,NL,KH)
      READ(8,100) (STI(I),I=1,NL,KH)
      READ(8,100) (NTI(I),I=1,NL,KH)
      READ(8,100) (NO3I(I),I=1,NL,KH)
      READ(8,100) (ALKI(I),I=1,NL,KH)
      IF(TYPE.EQ.3) GO TO 117
  115 DO 8 I=1,NL,KH
      KL=(I-1)/KH
      DUM=SIN(2*PI*(KL-8)/24)
      Q(1,I)=Q0*(1+VARQ*DUM)
      STI(I)=STI0*(1+VARSTI*DUM)
      NTI(I)=NTI0*(1+VARNTI*DUM)
      NO3I(I)=NO3I0*(1+VNO3I*DUM)
      ALKI(I)=ALKI0*(1+VALKI*DUM)
    8 CONTINUE
C
C        INTERPOLATION OF STI, NO3I, Q AND NTI AT REQUIRED INTERVALS
C        ------------------------------------------------------------
C
  117 DO 25 I=1,24
      KI=KH*I+1
      KJ=KH*(I-1)+1
      DIFSTI=(STI(KI)-STI(KJ))/KH
      DIFFQ=(Q(1,KI)-Q(1,KJ))/KH
      DIFNTI=(NTI(KI)-NTI(KJ))/KH
      DNO3I=(NO3I(KJ)-NO3I(KJ))/KH
      DALKI=(ALKI(KI)-ALKI(KJ))/KH
      DO 25 J=2,KH,1
      KK=KJ+J-1
      STI(KK)=STI(KJ)+(J-1)*DIFSTI
      Q(1,KK)=Q(1,KJ)+(J-1)*DIFFQ
      NTI(KK)=NTI(KJ)+(J-1)*DIFNTI
      NO3I(KK)=NO3I(KJ)+(J-1)*DNO3I
      ALKI(KK)=ALKI(KJ)+(J-1)*DALKI
   25 CONTINUE
      GO TO 118
  116 PQ=BQ+(VF-BQ)*24.0/LFP
      DO 24 I=1,NL,1
      NTI(I)=NTI00
      STI(I)=STI00
      Q(1,I)=BQ
```

```
512            ALKI(I)=ALKI00
513          24 NO3I(I)=NO3I00
514            NM=START*KH+1
515            END=START+LPP
516            NN=LPP*KH+NM
517            DO 23 I=NM,NN,1
518            STI(I)=STI0
519            Q(1,1)=PQ
520            NTI(I)=NTI0
521            NO3I(I)=NO3I0
522            ALKI(I)=ALKI0
523          23 CONTINUE
524         118 DO 1 I=1,NL
525            STQ(I)=STI(I)*Q(1,I)
526            NTQ(I)=NTI(I)*Q(1,I)
527            NO3Q(I)=NO3I(I)*Q(1,1)
528            ALKQ(I)=ALKI(I)*Q(1,I)
529            SUI(I)=FUS*STI(I)
530            NUI(I)=UN*NTI(I)
531            XII(I)=FUP*STI(I)
532            NXII(I)=XII(I)*PN
533            NH3I(I)=SN*NTI(I)
534            NONI(I)=NTI(I)-NUI(I)-NXII(I)-NH3I(I)
535          1 SBI(I)=STI(I)*(1.0-FUS-FUP*P)
536      C
537      C     READ IN STARTING VALUES OF THE VARIABLES
538      C
539      C
540            READ(8,100)(SBP(J,1),J=1,NR,1)
541            READ(8,100)(SBS(J,1),J=1,NR,1)
542            READ(8,100)(SU(J,1),J=1,NR,1)
543            READ(8,100)(XS(J,1),J=1,NR,1)
544            READ(8,100)(XA(J,1),J=1,NR,1)
545            READ(8,100)(XE(J,1),J=1,NR,1)
546            READ(8,100)(XI(J,1),J=1,NR,1)
547            READ(8,100)(XN(J,1),J=1,NR,1)
548            READ(8,100)(NU(J,1),J=1,NR,1)
549            READ(8,100)(NON(J,1),J=1,NR,1)
550            READ(8,100)(NH3(J,1),J=1,NR,1)
551            READ(8,100)(NO3(J,1),J=1,NR,1)
552            READ(8,100)(ALK(J,1),J=1,NR,1)
553            DO 46 J=1,NR,1
554            XV(J,1)=XS(J,1)+XA(J,1)+XE(J,1)+XI(J,1)
555      46    CONTINUE
556      C
557      C      CALCULATION OF AVERAGE SEWAGE CHARACTERISTICS
558      C      ---------------------------------------------
559      C
560            SUMQ=0
561            SUMSTI=0
562            SUMNTI=0
563            SNO3I=0.0
564            SALKI=0.0
565            QMAX=0.0
566            STIMAX=0.0
567            SUMSTQ=0.0
568            STQMAX=0.0
569            SUMNTQ=0.0
570            NTQMAX=0.0
571            NTIMAX=0.0
572            SNO3Q=0.0
573            NO3QAX=0.0
574            NO3IAX=0.0
575            SALKQ=0.0
```

```
576             ALKQAX=0.0
577             ALKIAX=0.0
578             DO 2 I=1,D,1
579             SUMNTI=SUMNTI+NTI(I)
580             SNO3I=SNO3I+NO3I(I)
581             SALKI=SALKI+ALKI(I)
582             SUMSTI=SUMSTI+STI(I)
583             SUMSTQ=SUMSTQ+STQ(I)
584             SUMNTQ=SUMNTQ+NTQ(I)
585             SNO3Q=SNO3Q+NO3Q(I)
586             SALKQ=SALKQ+ALKQ(I)
587             IF(STQ(I).GT.STQMAX) STQMAX=STQ(I)
588             IF(NTQ(I).GT.NTQMAX) NTQMAX=NTQ(I)
589             IF(NO3Q(I).GT.NO3QAX) NO3QAX=NO3Q(I)
590             IF(ALKQ(I).GT.ALKQAX) ALKQAX=ALKQ(I)
591             IF(Q(1,I).GT.QMAX) QMAX=Q(1,I)
592             IF(STI(I).GT.STIMAX) STIMAX=STI(1)
593             IF(NTI(I).GT.NTIMAX) NTIMAX=NTI(I)
594             IF(NO3I(I).GT.NO3IAX) NO3IAX=NO3I(I)
595             IF(ALKI(I).GT.ALKIAX) ALKIAX=ALKI(I)
596           2 SUMQ=SUMQ+Q(1,I)
597             QAV(1)=SUMQ/D
598             STQAV=SUMSTQ/D
599             NTQAV=SUMNTQ/D
600             NO3QAV=SNO3Q/D
601             ALKQAV=SALKQ/D
602             IF(TYPE.EQ.2) GO TO 119
603             NTIAV=SUMNTI/D
604             STIAV=SUMSTI/D
605             NO3IAV=SNO3I/D
606             ALKIAV=SALKI/D
607             GO TO 120
608         119 STIAV=STQAV/VF
609             NTIAV=NTQAV/VF
610             NO3IAV=NO3QAV/VF
611             ALKIAV=ALKQAV/VF
612         120 SBIAV=STIAV*(1.0-FUS-FUP*P)
613             SUIAV=FUS*STIAV
614             NUIAV=NTIAV*UN
615             XIIAV=FUP*STIAV
616             NXIIAV=FN*XIIAV
617             NH3IAV=SN*NTIAV
618             NONIAV=NTIAV-NXIIAV-NUIAV-NH3IAV
619             QSR=SR*QAV(1)
620             QAR=AR*QAV(1)
621             QBR=BR*QAV(1)
622             IF(TYPE.EQ.2) GO TO 121
623             VARSTL=STQMAX/STQAV-1.00
624             VARNTL=NTQMAX/NTQAV-1.00
625             VRNO3L=NO3QAX/NO3QAV-1.00
626             IF(VRALKL.LT.0.0) VRALKL=0.0
627             IF(VRNO3L.LT.0.0) VRNO3L=0.0
628             IF(VRALKL.LT.0.0) VRALKL=0.0
629             IF(TYPE.EQ.1) GO TO 121
630             VARQ=QMAX/QAV(1)-1.00
631             VARSTI=STIMAX/STIAV-1.00
632             VARNTI=NTIMAX/NTIAV-1.00
633             VNO3I=NO3IAX/NO3IAV-1.00
634             VALKI=ALKIAX/ALKIAV-1.00
635             IF(VNO3I.LT.0.0) VNO3I=0.0
636             IF(VALKI.LT.0.0) VALKI=0.0
637         121 CONTINUE
638       C
639       C             CALCULATE TOTAL AND AEROBIC VOLUMES
```

```
640       C              ------------------------------------
641       C
642             DO 37 J=1,NR,1
643             IFREQ=24/(ANOX(J)+AROB(J))
644             DO 37 I=1,NL,1
645             NTDN(J,I)=1
646       37    CONTINUE
647             VP=0.0
648             DO 133 J=1,NR,1
649             VP=VP+V(J)
650             DO 133 I=1,NL,1
651             DO 133 II=1,IFREQ,1
652             IB=(INIT(J)+(II-1)*(ANOX(J)+AROB(J)))*KH+1
653             IE=(INIT(J)+(II-1)*(ANOX(J)+AROB(J))+ANOX(J))*KH+1
654             IF(I.GE.IB.AND.I.LT.IE) NTDN(J,I)=0
655             NTDN(J,NL)=NTDN(J,1)
656         133 CONTINUE
657             JJ=3
658             DO 77 J=1,NR,1
659             INTDN(J)=0
660             DO 77 I=1,NL,1
661             INTDN(J)=INTDN(J)+NTDN(J,I)
662       77    CONTINUE
663             SANOX=0.0
664             DO 78 J=1,NR,1
665             IF(INTDN(J).EQ.0) JJ=1
666             IF(INTDN(J).EQ.NL) JJ=2
667             NAME(J)=LNAME(JJ)
668             FANOX(J)=1.0-INTDN(J)/NL
669             SANOX=SANOX+FANOX(J)
670       78    CONTINUE
671             LNAME(1)='AROB'
672             LNAME(2)='ANOX'
673             LNAME(3)='ANAE'
674       C
675       C                   ADJUST TEMPERATURE AND PH DEPENDENT CONSTANTS
676       C                   ---------------------------------------------
677       C
678             IF(PH.LT.8.0) UM20=UMAX
679             IF(PH.LT.7.2) UM20=UMAX*(PHIN)**(PH-7.20)
680             IF(PH.LT.8.0) KN20=KNMAX
681             IF(PH.LT.7.2) KN20=KNMAX*(PHIN)**(7.20-PH)
682             UMT=UM20*(THEN)**(TEMP-20.0)
683             KNT=KN20*(THEN)**(TEMP-20.0)
684             KSPT=KSP20*(THES)**(20.0-TEMP)
685             KMPT=KMP20*(THEM)**(TEMP-20.0)
686             KSST=KSS20*(THEZ)**(TEMP-20.0)
687             KMST=KMS20*(THEA)**(TEMP-20.0)
688             KRT=KR20*(THEE)**(TEMP-20.0)
689             KVT=KV20*(THEE)**(TEMP-20.0)
690             BHT=BH20*(THEE)**(TEMP-20.0)
691             BNT=BN20*(THEE)**(TEMP-20.0)
692             RSM=1.0/((UMT/(1.0+(KNT/(SN*NTIAV))))-BNT)
693             RSA=(1.0-SANOX)*RS
694             TRN=VP/QAV(1)
695             FNS=NONIAV/SBIAV
696             FACTC=STQAV/(STIAV*QAV(1))
697             FACTN=NTQAV/(NTIAV*QAV(1))
698             CC=(1.0-F)*BHT*YH*RS/(1.0+BHT*(1.0-YH*P*(1.0-F))*RS)
699             FNS=(FNS*(1.0-FCA)+FN*CC)/((1.0-FCA)+P*CC)
700       C
701       C                CALCULATE INTEGRTION STEP LENGTHS
702       C                ---------------------------------
703             DT=1./D
```

```
      DDT=1./DD
      RDT=DDT/DT
C
C           RELATE STEP INTERVALS TO THE TIME BASE
C           --------------------------------------
C
      II=0
      KN=M/D
      KH1=KH+1
      DO 27 I=1,NL,1
      JE=I-1
      IF(JE.EQ.0) GO TO 3
      KO(I)=KO(I-1)+KN
      GO TO 4
    3 KO(I)=KOM
    4 II=II+1
      IF (II.EQ.KH1) KO(I)=KO(I)+40
      IF (II.EQ.KH1) II=1
      IF (KO(I).GE.2400) KO(I)=KO(I)-2400
      IF(KO(I).EQ.ICSD) IC=I-1
      IF(KOM.EQ.ICSD) IC=1
      IF(KO(I).EQ.ISSD) IS=I
   27 CONTINUE
      HK=IS-IC
      DO 9 J=1,NR,1
      DO 29 I=1,NL,1
      WQ(J,I)=0.0
   29 CONTINUE
      IF(LRSD(J).EQ.0) GO TO 32
      DO 28 I=IC,IS,1
      WQ(J,I)=VP/RS*D/HK*1.0/NRSD
   28 CONTINUE
   32 DO 48 I=1,NL,1
      Q(J+1,I)=Q(J,I)-WQ(J,I)
      IF(Q(J+1,I).GT.0.00) GO TO 48
      Q(J+1,I)=0.00
      WQ(J,I)=Q(J,I)
   48 CONTINUE
    9 CONTINUE
C
C           SET INITIAL CONDITIONS
C           ----------------------
C
      DUM=(QSR+Q(NRR,1))/QSR
      SBPR(1)=SBP(NR,1)*DUM
      XSR(1)=XS(NR,1)*DUM
      XAR(1)=XA(NR,1)*DUM
      XER(1)=XE(NR,1)*DUM
      XIR(1)=XI(NR,1)*DUM
      XNR(1)=XN(NR,1)*DUM
      DO 5 J=1,NR,1
      WRITE(5,200) SBS(J,1),SBP(J,1),XS(J,1),XA(J,1),XE(J,1),XI(J,1),
     1XV(J,1),XN(J,1),NO3(J,1),ALK(J,1),SU(J,1),NON(J,1),NH3(J,1)
    5 CONTINUE
  200 FORMAT(1H ,5X,13F7.2)
      DO 15 I=1,D,1
   17 J=0
   19 J=J+1
C*****************************************************************
      IF(NTDN(J,I).EQ.1) CALL UAERO(D,DT,J,FR,FCA,FCS,KVT,KMPT,KSPT,
     1KMST,KSST,P,NR,YH,BHT,F,KRT,FOE,FN,FOS,FNS,NJ,DDT,UMT,YN,KNT,
     2RDT,BNT,FACT,LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,I)
C*****************************************************************
      IF(NTDN(J,I).EQ.0) CALL UANOX(D,DT,J,FR,FCA,FCS,KVT,KMPT,KSPT,
```

```
768           1KMST,KSST,P,NR,YH,BHT,F,KRT,FOE,FN,FOS,FNS,NJ,DDT,UMT,YN,KNT,
769           2RDT,BNT,FACT,LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,I)
770     C*****************************************************************
771           IF(J.EQ.NR) GO TO 20
772           GO TO 19
773        20 CONTINUE
774     C*****************************************************************
775           CALL USETL(NL,J,QSR,I)
776     C*****************************************************************
777        15 CONTINUE
778           DO 26 J=1,NR,1
779           DO 26 I=1,NL
780           JA=I-1
781           IF (JA.EQ.0) JA=D
782           JB=I+1
783           IF (JB.EQ.NLL) JB=2
784           JD=I
785           IF(JD.EQ.NL) JD=1
786           RXS(J,I)=(XS(J,JB)-XS(J,JA))/(DT*2.0)
787           ST(J,I)=SU(J,I)+SBS(J,I)
788           NT(J,I)=NON(J,I)+NU(J,I)+NH3(J,I)
789           OC(J,I)=(1.0-P*YH)*SYNTH(J,JD)/(DT*24.0)
790           ANPOT(J,I)=ANPOT(J,JD)/(DT*24.0)
791           ON(J,I)=4.57*NIT(J,JD)/(DT*24.0)
792           OT(J,I)=OC(J,I)+ON(J,I)
793           AMON(J,I)=AMON(J,JD)/(DT*24.0)
794        26 CONTINUE
795     C
796     C       CALCULATION OF AVERAGE RESULTS FROM DYNAMIC MODEL
797     C       ------------------------------------------------
798     C           EXAMPLE:  XVAV = AVERAGE XV
799     C
800           TRN=0.0
801           SUMXVT=0.0
802           SUMWST=0.0
803           SUMXAM=0.0
804           SUMXEM=0.0
805           SUMX1M=0.0
806           SUMXSM=0.0
807           SUMXNM=0.0
808           SUMODM=0.0
809           SMAPTM=0.0
810           APTMNR=0.0
811           RNO3AV=0.0
812           SUMOCM=0.0
813           SUMONM=0.0
814           SUMOTM=0.0
815           SUMAMM=0.0
816           SUNSWM=0.0
817           TWCOD=0.0
818           TWN=0.0
819           TWALK=0.0
820           EN=0.0
821           ENH3=0.0
822           ENON=0.0
823           ENO3=0.0
824           EALK=0.0
825           ENT=0.0
826           ECOD=0.0
827           DO 31 J=2,NRR,1
828           SUMQ=0.0
829           DO 30 I=1,D,1
830        30 SUMQ=SUMQ+Q(J,I)
831        31 QAV(J)=SUMQ/D
```

```
832             DO 22 J=1,NR,1
833             SUMWS=0.0
834             SUMNSW=0.0
835             SUMNU=0.0
836             SUMXA=0.0
837             SUMXE=0.0
838             SUMXI=0.0
839             SUMSBP=0.0
840             SUMSBS=0.0
841             SUMXV=0.0
842             SUMXS=0.0
843             SUMXN=0.0
844             SUMOC=0.0
845             SUMOD=0.0
846             SUMAPT=0.0
847             SUMON=0.0
848             SUMSU=0.0
849             SUMRXS=0.0
850             SUMNON=0.0
851             SUMNH3=0.0
852             SUMNT=0.0
853             SUMNO3=0.0
854             SUMALK=0.0
855             SUMOT=0.0
856             SUMAM=0.0
857             WCOD=0.0
858             WN=0.0
859             WALK=0.0
860             WNON=0.0
861             DO 21 I=1,D
862             SUMWS=SUMWS+XV(J,I)*WQ(J,I)*DT
863             SUMNSW=SUMNSW+(FN*(XA(J,I)+XE(J,I)+XI(J,I))+FNS*XS(J,I))
864            1*WQ(J,I)*DT
865             IF(J.NE.NR) GO TO 11
866             ECOD=ECOD+Q(J+1,I)*ST(J,I)*DT/QAV(J+1)
867             ESBS=ESBS+Q(J+1,I)*(SBS(J,I))*DT/QAV(J+1)
868             ESU=ESU+Q(J+1,I)*(SU(J,I))*DT/QAV(J+1)
869             EN=EN+Q(J+1,I)*(NT(J,I)+NO3(J,I))*DT/QAV(J+1)
870             ENH3=ENH3+Q(J+1,I)*(NH3(J,I))*DT/QAV(J+1)
871             ENON=ENON+Q(J+1,I)*(NON(J,I))*DT/QAV(J+1)
872             ENO3=ENO3+Q(J+1,I)*(NO3(J,I))*DT/QAV(J+1)
873             EALK=EALK+Q(J+1,I)*(ALK(J,I))*DT/QAV(J+1)
874             ENT=ENT+Q(J+1,I)*(NT(J,I))*DT/QAV(J+1)
875          11 SUMNU=SUMNU+NU(J,I)
876             SUMXA=SUMXA+XA(J,I)
877             SUMXE=SUMXE+XE(J,I)
878             SUMXI=SUMXI+XI(J,I)
879             SUMSBP=SUMSBP+SBP(J,I)
880             SUMSBS=SUMSBS+SBS(J,I)
881             SUMXV=SUMXV+XV(J,I)
882             SUMXN=SUMXN+XN(J,I)
883             SUMXS=SUMXS+XS(J,I)
884             SUMOC=SUMOC+OC(J,I)
885             SUMAPT=SUMAPT+ANPOT(J,I)
886             IF(NTDN(J,I).EQ.1) GO TO 12
887             SUMOD=SUMOD+OC(J,I)
888          12 SUMON=SUMON+ON(J,I)
889             SUMSU=SUMSU+SU(J,I)
890             SUMRXS=SUMRXS+RXS(J,I)
891             SUMNO3=SUMNO3+NO3(J,I)
892             SUMALK=SUMALK+ALK(J,I)
893             SUMNON=SUMNON+NON(J,I)
894             SUMNH3=SUMNH3+NH3(J,I)
895             SUMNT=SUMNT+NT(J,I)
```

```
      WCOD=WCOD+WQ(J,I)*(ST(J,I)+SBP(J,I))*DT
      WN=WN+WQ(J,I)*(NT(J,I)+NO3(J,I))*DT
      WALK=WALK+WQ(J,1)*ALK(J,I)*DT
      WNON=WNON+WQ(J,1)*NON(J,I)*DT
      SUMOT=SUMOT+OT(J,I)
      SUMAM=SUMAM+AMON(J,I)
   21 CONTINUE
      NUAV(J)=SUMNU/D
      XAAV(J)=SUMXA/D
      XEAV(J)=SUMXE/D
      XIAV(J)=SUMXI/D
      SBPAV(J)=SUMSBP/D
      SBSAV(J)=SUMSBS/D
      XVAV(J)=SUMXV/D
      XSAV(J)=SUMXS/D
      XNAV(J)=SUMXN/D
      OCAV(J)=SUMOC/D
      ODAV(J)=SUMOD/D
      APTAV(J)=SUMAPT/D
      APTNR(J)=APTAV(J)*24.0*V(J)/QAV(1)
      RNO3(J)=ODAV(J)*24.0*V(J)/(QAV(1)*2.85)
      ONAV(J)=SUMON/D
      SUAV(J)=SUMSU/D
      RXSAV(J)=SUMRXS/D
      STAV(J)=SUAV(J)+SBSAV(J)
      NSWAV(J)=SUMNSW/QAV(1)
      SUNSWM=SUNSWM+NSWAV(J)
      OTAV(J)=SUMOT/D
      AMAV(J)=SUMAM/D
      NO3AV(J)=SUMNO3/D
      ALKAV(J)=SUMALK/D
      NONAV(J)=SUMNON/D
      NH3AV(J)=SUMNH3/D
      NTAV(J)=SUMNT/D
      SUMWST=SUMWST+SUMWS
      SUMXVT=SUMXVT+XVAV(J)*V(J)
      SUMXAM=SUMXAM+XAAV(J)*V(J)
      SUMXEM=SUMXEM+XEAV(J)*V(J)
      SUMXIM=SUMXIM+XIAV(J)*V(J)
      SUMXSM=SUMXSM+XSAV(J)*V(J)
      SUMXNM=SUMXNM+XNAV(J)*V(J)
      SUMODM=SUMODM+ODAV(J)*V(J)
      SMAPTM=SMAPTM+APTAV(J)*V(J)
      APTMNR=APTMNR+APTNR(J)
      RNO3AV=RNO3AV+RNO3(J)
      SUMOCM=SUMOCM+OCAV(J)*V(J)
      SUMONM=SUMONM+ONAV(J)*V(J)
      SUMOTM=SUMOTM+OTAV(J)*V(J)
      SUMAMM=SUMAMM+AMAV(J)*V(J)
      I=NLL
      Q(J,I)=QAV(J)
C*****************************************************************
      CALL UDUMV(J,I,DUMIF,DUMS,DUMA,DUMB,DUM1R,DUMO,
     1LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,
     1DT,DDT,RDT)
C*****************************************************************
      RA(J)=24.0*DDT/DUMO
      RN(J)=V(J)/QAV(J)*24.0
      TWCOD=TWCOD+WCOD
      TWN=TWN+WN
      TWALK=TWALK+WALK
      TWNON=TWNON+WNON
      TRN=TRN+RN(J)
      IF(NTDN(J,I).EQ.1) GO TO 22
```

```
960           SUMXVA=SUMXVA+XVAV(J)*V(J)
961        22 CONTINUE
962           RSAV=SUMXVT/SUMWST
963           XAMAV=SUMXAM/VP
964           XEMAV=SUMXEM/VP
965           XIMAV=SUMXIM/VP
966           XSMAV=SUMXSM/VP
967           XVMAV=SUMXVT/VP
968           XNMAV=SUMXNM/VP
969           ODMAV=SUMODM/VP
970           APTMAV=SMAPTM/VP
971           OCMAV=SUMOCM/VP
972           ONMAV=SUMONM/VP
973           OTMAV=SUMOTM/VP
974           AMMAV=SUMAMM/VP
975           CODSI=0.0
976           NTITI=0.0
977           CODSO=0.0
978           NTITO=0.0
979           ALKSI=0.0
980           ALKSO=0.0
981           DO 5000 J=1,NR,1
982           CODSI=CODSI+(XV(J,1)*P+SBP(J,1)+ST(J,1))*V(J)
983           NTITI=NTITI+((XA(J,1)+XE(J,1)+XI(J,1))*FN+XS(J,1)*FNS
984          &+NON(J,1)+NU(J,1)+NH3(J,1)+NO3(J,1))*V(J)
985           CODSO=CODSO+(XV(J,D)*P+SBP(J,D)+ST(J,D))*V(J)
986           NTITO=NTITO+((XA(J,D)+XE(J,D)+XI(J,D))*FN+XS(J,D)*FNS
987          &+NON(J,D)+NU(J,D)+NH3(J,D)+NO3(J,D))*V(J)
988           ALKSI=ALKSI+ALK(J,1)*V(J)
989           ALKSO=ALKSO+ALK(J,D)*V(J)
990     5000  CONTINUE
991     C
992     C     CHECK MASS BALANCES ON COD,NITROGEN AND ALKALINITY
993     C  1.  COD
994     C  -------
995           TCODI=STQAV+CODSI
996           TCODO=P*SUMWST+SUMOCM*24.0+TWCOD+ECOD*QAV(NRR)+CODSO
997           BALC=TCODO*100.0/TCODI
998     C  2.  NITROGEN
999     C  ------------
1000          TNTI=NTQAV+NO3QAV+NTITI
1001          TNTO=SUNSWM*QAV(1)+TWN+EN*QAV(NRR)+SUMODM*24/2.85+NTITO
1002          BALN=TNTO*100.0/TNTI
1003    C  3.   ALKALINITY
1004    C  ---------------
1005          TALKI=ALKQAV+ALKSI
1006          TALKO=TWALK+EALK*QAV(NRR)+ALKSO+7.14*SUMONM*24.0/4.57
1007         &-3.57*(SUMODM*24.0/2.86+SUMAMM*24.0)
1008          BALALK=TALKO/TALKI*100.0
1009    C
1010    C
1011    C
1012    C      OUTPUT AND FORMAT STATEMENTS
1013    C      ----------------------------
1014    C
1015          IF(RESULT.EQ.2)GO TO 126
1016          WRITE(5,98)DATE,XMONTH,YEAR
1017    98    FORMAT(1H0,5X,'DAY OF EXPERIMENT:',1X,A2,1X,'OF',A3,1X,A4)
1018          WRITE(5,201)TEMP,PH,YH,BHT,KVT,YN,THEN,F,FCA,FN,KNT,PHIN,
1019         1THEE,P,FOE,UMT,BNT,FOS,KRT,KSPT,THEZ,KMPT,THEA,THES,
1020         2KMST,KSST,FR,FNS,RS
1021       201 FORMAT(1H1,20X,22('*'),/,1H ,20X,22('*'),/,
1022          11H0,10X,'KINETIC CONSTANTS',/,1H ,10X,17('-'),/,
1023          21H ,5X,'PROCESS OPERATING TEMPERATURE=',F5.1,1X,'DEG. C',/,
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1024          31H ,28X,'AND PH=',F5.2,/,1H0,5X,'HETEROTROPHS',28X,'AUTOTROPHS',/,
1025          41H ,5X,12('-'),28X,10('-'),/,1H ,6X,'YH=',F5.3,4X,'BH=',F5.3,4X,
1026          5'KV=',F5.3,8X,'YN=',F5.3,6X,'THEN=',F5.3,/,1H ,7X,'F=',F5.3,3X,
1027          6'FCA=',F5.3,4X,'FN=',F5.3,8X,'KN=',F5.2,6X,'PHIN=',F5.3,/,
1028          75X,'THEE=',F5.3,5X,'P=',F5.3,3X,'FOE=',F5.3,8X,'UM=',F5.3,
1029          28X,'BN=',F5.3,/,
1030          86X,'FOS=',F5.3,4X,'KR=',F5.3,3X,'KSP=',F5.1,6X,'THEZ=',F5.3,/,
1031          16X,'KMP=',F5.3,2X,'THEA=',F5.3,2X,'THES=',F5.3,/,
1032          26X,'KMS=',F5.2,3X,'KSS=',F5.1,/,
1033          27X,'FR=',F5.3,3X,'FNS=',F5.3,/,
1034          11H ,5X,'OPERATING SLUDGE AGE OF PROCESS=',F5.2,1X,'DAYS')
1035           WRITE(5,202)FUP,FUS,SN,UN,QAV(1),STQAV,NTQAV,STIAV,SBIAV,SUIAV,
1036          1XIIAV,NTIAV,NH3IAV,NONIAV,NUIAV,NO3IAV,ALKIAV
1037        202 FORMAT(1H0,10X,'AVERAGE COMPOSITION OF SEWAGE',/,
1038          11H ,10X,29('-'),/,1H0,5X,'SOLID INERT FRACTION OF INFLUENT COD=',
1039          2F5.3,/,1H ,5X,'SOLUBLE INERT FRACTION OF INFLUENT COD=',F5.3,/,
1040          31H ,5X,'AMMONIA TO TKN FRACTION=',F5.3,/,
1041          41H ,5X,'INERT TKN FRACTION=',F5.3,/,1H0,4X,'QAV=',F9.5,
1042          15X,'STQAV=',F9.5,5X,'NTQAV=',F9.5,/,
1043          51H ,5X,'STI=',F7.2,5X,'SBI=',F7.2,4X,'SUI=',F6.2,
1044          64X,'XII=',F6.2,/,1H ,5X,'NTI=',F6.2,4X,'NH3I=',F6.2,
1045          74X,'NONI=',F5.2,5X,'NUI=',F5.2,5X,'NO3I=',F5.2,/,
1046          &4X,'ALKI=',F7.2)
1047           IF(TYPE.EQ.1) GO TO 122
1048           IF(TYPE.EQ.2) GO TO 123
1049           IF(TYPE.EQ.3) GO TO 122
1050        122 WRITE(5,205)VARQ,VARSTI,VARNTI,VNO3I,VARSTL,VARNTL,VRNO3L
1051        205 FORMAT(1H0,10X,'SINE WAVE INFLUENT FLOW AND FEED PATTERN',/,
1052          11H ,10X,40('-'),/,
1053          41H0,10X,'AMPLITUDE OF FLOW WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1054          51H ,10X,'AMPLITUDE OF COD WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1055          61H ,10X,'AMPLITUDE OF TKN WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1056          71H ,10X,'AMPLITUDE OF NO3 WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1057          81H ,10X,'AMPLITUDE OF COD LOAD WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1058          81H ,10X,'AMPLITUDE OF TKN LOAD WAVE=',F5.2,4X,'(FRAC OF AVE)',/,
1059          81H ,10X,'AMPLITUDE OF NO3 LOAD WAVE=',F5.2,4X,'(FRAC OF AVE)')
1060        204 FORMAT(1H0,10X,'PROCESS CONFIGURATION DATA',/,1H ,5X,26('-'),/,
1061          11H0,5X,'NUMBER OF TANKS IN SERIES',I2,/,
1062          11H ,5X,'INFLUENT FLOW INTO TANK',I2,/,
1063          11H ,5X,'S-RECYCLE FROM SETTLER TO TANK ',I2,/,
1064          11H ,5X,'A-RECYCLE FROM TANK',I2,1X,'INTO TANK',I2,/,
1065          11H ,5X,'B-RECYCLE FROM TANK',I2,1X,'INTO TANK',I2,/,
1066          11H ,5X,'S RECYCLE RATIO=',F4.1,/,
1067          11H ,5X,'A RECYCLE RATIO=',F4.1,/,
1068          11H ,5X,'B RECYCLE RATIO=',F4.1)
1069           GO TO 124
1070        123   WRITE(5,206)LFP,PQ,BQ,VF
1071        206 FORMAT(1H0,10X,'SQUARE WAVE INFLUENT FLOW AND FEED PATTERN',/,
1072          11H ,10X,42('-'),/,
1073          11H ,10X,'LENGTH OF FEED PERIOD (HRS)=',F5.2,/,
1074          21H ,10X,'PEAK FLOW (L/D)=',F9.5,/,
1075          31H ,10X,'BASE FLOW (L/D)=',F9.5,/,
1076          41H ,10X,'VOLUME OF FEED PER DAY (LITRES)=' F9.5)
1077        124 WRITE(5,203) TRN,VP,ANOXAV,RSAV
1078           WRITE(5,204) NR,LINI,LSRI,LARO,LARI,LBRO,LBRI,SR,AR,BR
1079        203 FORMAT(1H1,10X,21('-'),/,1H ,10X,'DAILY AVERAGE RESULTS',/,
1080          11H ,10X,21('-'),/,
1081          11H0,5X,'TOTAL HYDRAULIC RETENTION TIME=',F8.3,1X,'HOURS',/,
1082          11H ,5X,'TOTAL VOLUME=',F9.5,1X,'LITRES',/,
1083          11H ,5X,'ANOXIC FRACTION=',F5.3,/,
1084          21H ,5X,'SLUDGE AGE =',F5.2,1X,'DAYS')
1085           WRITE(5,244)TCODI,TCODO,BALC,TNTI,TNTO,BALN,TALKI,TALKO,BALALK
1086        244 FORMAT(1H0,10X,'MASS BALANCES ON COD AND NITROGEN',/,
1087          11H ,10X,33('-'),/,
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1088          21H0,5X,'TOTAL COD MASS INPUT=',1X,E13.6,1X,'(MG-COD/D)',/,
1089          21H ,5X,'TOTAL COD MASS OUTPUT=',1X,E13.6,1X,'(MG-COD/D)',/,
1090          31H ,5X,'PERCENTAGE COD RECOVERY=',1X,F7.3,1X,'(%)',/,
1091          21H ,5X,'TOTAL NITROGEN MASS INPUT=',1X,E13.6,1X,'(MG-N/D)',/,
1092          21H ,5X,'TOTAL NITROGEN MASS OUTPUT=',1X,E13.6,1X,'(MG-N/D)',/,
1093          31H ,5X,'PERCENTAGE NITROGEN RECOVERY=',1X,F7.3,1X,'(%)',/,
1094          21H ,5X,'TOTAL ALKALINITY MASS INPUT=',1X,E13.6,1X,'(CACO3/D)',/,
1095          21H ,5X,'TOTAL ALKALINITY MASS OUTPUT=',1X,E13.6,1X,'(CACO3/D)',/,
1096          31H ,5X,'PERCENTAGE ALKALINITY RECOVERY=',1X,F7.3,1X,'(%)')
1097           WRITE(5,300)
1098        300 FORMAT(1H0,30X,'TANK  1',6X,'TANK  2',
1099           26X,'TANK  3',6X,'TANK  4',6X,'TANK  5',6X,'TANK  6',
1100           36X,'MEAN',/,1H ,30X,7('-'),6X,7('-'),6X,7('-'),6X,7('-'),
1101           46X,7('-'),6X,7('-'),6X,4('-'))
1102           WRITE(5,327) (NAME(J),J=1,NR,1)
1103        327 FORMAT(1H ,23X,6A13)
1104           WRITE(5,328)
1105        328 FORMAT(1H ,30X,6('-'),7X,6('-'),7X,6('-'),
1106           17X,6('-'),7X,6('-'),7X,6('-'))
1107           WRITE(5,301) (XAAV(J),J=1,NR,1),XAMAV
1108           WRITE(5,302) (XSAV(J),J=1,NR,1),XSMAV
1109           WRITE(5,303) (XEAV(J),J=1,NR,1),XEMAV
1110           WRITE(5,304) (XIAV(J),J=1,NR,1),XIMAV
1111           WRITE(5,305) (XVAV(J),J=1,NR,1),XVMAV
1112           WRITE(5,319) (XNAV(J),J=1,NR,1),XNMAV
1113           WRITE(5,313) (ODAV(J),J=1,NR,1),ODMAV
1114           WRITE(5,314) (OCAV(J),J=1,NR,1),OCMAV
1115           WRITE(5,315) (ONAV(J),J=1,NR,1),ONMAV
1116           WRITE(5,316) (OTAV(J),J=1,NR,1),OTMAV
1117           WRITE(5,3160) (AMAV(J),J=1,NR,1),AMMAV
1118           WRITE(5,309) (NSWAV(J),J=1,NR,1),SUNSWM
1119           WRITE(5,321) (SBPAV(J),J=1,NR,1)
1120           WRITE(5,320) (SBSAV(J),J=1,NR,1)
1121           WRITE(5,324) (SUAV(J),J=1,NR,1)
1122           WRITE(5,311) (STAV(J),J=1,NR,1)
1123           WRITE(5,306) (NONAV(J),J=1,NR,1)
1124           WRITE(5,310) (NH3AV(J),J=1,NR,1)
1125           WRITE(5,323) (NUAV(J),J=1,NR,1)
1126           WRITE(5,312) (NTAV(J),J=1,NR,1)
1127           WRITE(5,307) (NO3AV(J),J=1,NR,1)
1128           WRITE(5,703) (ALKAV(J),J=1,NR,1)
1129           WRITE(5,308) (RNO3(J),J=1,NR,1),RNO3AV
1130           WRITE(5,326) (APTAV(J),J=1,NR,1),APTMAV
1131           WRITE(5,329) (APTNR(J),J=1,NR,1),APTMNR
1132           WRITE(5,322) (V(J),J=1,NR,1)
1133           WRITE(5,325) (QAV(J),J=1,NRR,1)
1134           WRITE(5,317) (RA(J),J=1,NR,1)
1135           WRITE(5,318) (RN(J),J=1,NR,1)
1136           WRITE(5,3003)(FANOX(J),J=1,NR,1),ANOXAV
1137           WRITE(5,3000)ECOD,ESBS,ESU,EN,ENT,ENH3,ENON,ENO3,EALK
1138      3000  FORMAT(1H0,8X,'AVERAGE EFFLUENT CONCENTRATIONS',/,
1139          &8X,'TOT-COD=',E13.5,/,
1140          &8X,'SBS',E13.5,/,
1141          &8X,'SU',E13.5,/,
1142          &8X,'TOTAL-N=',E13.5,/,
1143          &8X,'TKN',E13.5,/,
1144          &8X,'NH3',E13.5,/,
1145          &8X,'NON',E13.5,/,
1146          &8X,'NO3',E13.5,/,
1147          &8X,'ALK',E13.5)
1148        301 FORMAT(1H ,8X,'XA    (MG-VSS/L)',2X,7(E13.5))
1149        302 FORMAT(1H ,8X,'XS    (MG-VSS/L)',2X,7(E13.5))
1150        303 FORMAT(1H ,8X,'XE    (MG-VSS/L)',2X,7(E13.5))
1151        304 FORMAT(1H ,8X,'XI    (MG-VSS/L)',2X,7(E13.5))
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1152        305 FORMAT(1H ,8X,'XV    (MG-VSS/L)',2X,7(E13.5))
1153        319 FORMAT(1H ,8X,'XN    (MG-VSS/L)',2X,7(E13.5))
1154        313 FORMAT(1H ,8X,'OD   (MG-O/L/HR)',2X,7(E13.5))
1155        314 FORMAT(1H ,8X,'OC   (MG-O/L/HR)',2X,7(E13.5))
1156        315 FORMAT(1H ,8X,'ON   (MG-O/L/HR)',2X,7(E13.5))
1157        316 FORMAT(1H ,8X,'OT   (MG-O/L/HR)',2X,7(E13.5))
1158       3160 FORMAT(1H ,8X,'AM   (MG-N/L/HR)',2X,7(E13.5))
1159        309 FORMAT(1H ,8X,'NSW   (MG-N/L/D)',2X,7(E13.5))
1160        321 FORMAT(1H ,8X,'SBP   (MG-COD/L)',2X,6(E13.5))
1161        320 FORMAT(1H ,8X,'SBS   (MG-COD/L)',2X,6(E13.5))
1162        324 FORMAT(1H ,8X,'SU    (MG-COD/L)',2X,6(E13.5))
1163        311 FORMAT(1H ,8X,'ST    (MG-COD/L)',2X,6(E13.5))
1164        306 FORMAT(1H ,8X,'NON     (MG-N/L)',2X,6(E13.5))
1165        310 FORMAT(1H ,8X,'NH3     (MG-N/L)',2X,6(E13.5))
1166        323 FORMAT(1H ,8X,'NU      (MG-N/L)',2X,6(E13.5))
1167        312 FORMAT(1H ,8X,'NT      (MG-N/L)',2X,6(E13.5))
1168        307 FORMAT(1H ,8X,'NO3     (MG-N/L)',2X,6(E13.5))
1169        703 FORMAT(1H ,8X,'ALK (MG-CACO3/L)',2X,6(E13.5))
1170        326 FORMAT(1H ,8X,'APT (MG-N/L/HR)',2X,7(E13.5))
1171        329 FORMAT(1H ,8X,'APTNR   (MG-N/L)',2X,7(E13.5))
1172        308 FORMAT(1H ,8X,'RNO3    (MG-N/L)',2X,7(E13.5))
1173        322 FORMAT(1H ,8X,'VOL     (LITRES)',2X,6(E13.5))
1174        325 FORMAT(1H ,8X,'FLOW       (L/D)',2X,7(E13.5))
1175        317 FORMAT(1H ,8X,'AHRT     (HOURS)',2X,6(E13.5))
1176        318 FORMAT(1H ,8X,'NHRT     (HOURS)',2X,6(E13.5))
1177       3003 FORMAT(1H ,8X,'ANOX FR    (%)',2X,7(E13.5))
1178            WRITE(CH,'(I1)')NR
1179            IMFT(10:10)=CH
1180            WRITE(5,399) (KO(I),STQ(I),NTQ(I),Q(1,I),SBI(I),STI(I),
1181           1NTI(I),SUI(I),NUI(I),XII(I),NH3I(I),NONI(I),I=1,NL,KH)
1182        399 FORMAT(1H1,10X,'DYNAMIC INPUTS',/,1H ,10X,14('-'),/,
1183           11H ,2X,'TIME',2X,'COD LOAD',2X,'TKN LOAD',4X,'FLOW',7X,'SBI',7X,
1184           2'STI',7X,'NTI',7X,'SUI',7X,'NUI',7X,'XII',6X,'NH3I',6X,'NONI',/,
1185           31H ,2X,4('-'),2X,8('-'),3X,8('-'),4X,4('-'),7X,3('-'),7X,3('-'),
1186           47X,3('-'),7X,3('-'),7X,3('-'),7X,3('-'),6X,4('-'),6X,4('-'),/,
1187           5(1H ,2X,I4,11(E10.4)))
1188            WRITE(5,400)
1189        400 FORMAT(1H1,10X,21('-'),/,1H ,10X,'DAILY DYNAMIC RESULTS',/,
1190           11H ,10X,21('-'),/,
1191           11H0,20X,'TIME',5X,'TANK  1',6X,'TANK  2',
1192           26X,'TANK  3',6X,'TANK  4',6X,'TANK  5',6X,'TANK  6',/,
1193           31H ,20X,4('-'),5X,7('-'),6X,7('-'),6X,7('-'),6X,7('-'),
1194           46X,7('-'),6X,7('-'))
1195            WRITE(5,401)
1196        401 FORMAT(1H0,3X,'XA    (MG-VSS/L)')
1197            WRITE(5,IMFT) (KO(I),(XA(J,I),J=1,NR,1),I=1,NL,KH)
1198            WRITE(5,402)
1199        402 FORMAT(1H0,3X,'XS    (MG-VSS/L)')
1200            WRITE(5,IMFT) (KO(I),(XS(J,I),J=1,NR,1),I=1,NL,KH)
1201            WRITE(5,403)
1202        403 FORMAT(1H0,3X,'XE    (MG-VSS/L)')
1203            WRITE(5,IMFT) (KO(I),(XE(J,I),J=1,NR,1),I=1,NL,KH)
1204            WRITE(5,404)
1205        404 FORMAT(1H0,3X,'XI    (MG-VSS/L)')
1206            WRITE(5,IMFT) (KO(I),(XI(J,I),J=1,NR,1),I=1,NL,KH)
1207            WRITE(5,405)
1208        405 FORMAT(1H0,3X,'XV    (MG-VSS/L)')
1209            WRITE(5,IMFT) (KO(I),(XV(J,I),J=1,NR,1),I=1,NL,KH)
1210            WRITE(5,419)
1211        419 FORMAT(1H0,3X,'XN    (MG-VSS/L)')
1212            WRITE(5,IMFT) (KO(I),(XN(J,I),J=1,NR,1),I=1,NL,KH)
1213            WRITE(5,407)
1214        407 FORMAT(1H0,3X,'OC   (MG-O/L/HR)')
1215            WRITE(5,IMFT) (KO(I),(OC(J,I),J=1,NR,1),I=1,NL,KH)
```

```
1216            WRITE(5,408)
1217          408 FORMAT(1H0,3X,'ON  (MG-O/L/HR)')
1218            WRITE(5,IMFT) (KO(I),(ON(J,I),J=1,NR,1),I=1,NL,KH)
1219            WRITE(5,409)
1220          409 FORMAT(1H0,3X,'OT  (MG-O/L/HR)')
1221            WRITE(5,IMFT) (KO(I),(OT(J,I),J=1,NR,1),I=1,NL,KH)
1222            WRITE(5,4090)
1223         4090  FORMAT(1H0,3X,'AM (MG-N/L/HR)')
1224            WRITE(5,IMFT) (KO(I),(AMON(J,I),J=1,NR,1),I=1,NL,KH)
1225            WRITE(5,417)
1226          417 FORMAT(1H0,3X,'WQ       (L/DAY)')
1227            WRITE(5,IMFT) (KO(I),(WQ(J,I),J=1,NR,1),I=1,NL,KH)
1228            WRITE(5,411)
1229          411 FORMAT(1H0,3X,'SBP  (MG-COD/L)')
1230            WRITE(5,IMFT) (KO(I),(SBP(J,I),J=1,NR,1),I=1,NL,KH)
1231            WRITE(5,418)
1232          418 FORMAT(1H0,3X,'SBS  (MG-COD/L)')
1233            WRITE(5,IMFT) (KO(I),(SBS(J,I),J=1,NR,1),I=1,NL,KH)
1234            WRITE(5,420)
1235          420 FORMAT(1H0,3X,'SU   (MG-COD/L)')
1236            WRITE(5,IMFT) (KO(I),(SU(J,I),J=1,NR,1),I=1,NL,KH)
1237            WRITE(5,412)
1238          412 FORMAT(1H0,3X,'ST   (MG-COD/L)')
1239            WRITE(5,IMFT) (KO(I),(ST(J,I),J=1,NR,1),I=1,NL,KH)
1240            WRITE(5,413)
1241          413 FORMAT(1H0,3X,'NON    (MG-N/L)')
1242            WRITE(5,IMFT) (KO(I),(NON(J,I),J=1,NR,1),I=1,NL,KH)
1243            WRITE(5,421)
1244          421 FORMAT(1H0,3X,'NU     (MG-N/L)')
1245            WRITE(5,IMFT) (KO(I),(NU(J,I),J=1,NR,1),I=1,NL,KH)
1246            WRITE(5,415)
1247          415 FORMAT(1H0,3X,'NH3    (MG-N/L)')
1248            WRITE(5,IMFT) (KO(I),(NH3(J,I),J=1,NR,1),I=1,NL,KH)
1249            WRITE(5,414)
1250          414 FORMAT(1H0,3X,'NT     (MG-N/L)')
1251            WRITE(5,IMFT) (KO(I),(NT(J,I),J=1,NR,1),I=1,NL,KH)
1252            WRITE(5,422)
1253          422 FORMAT(1H0,3X,'APT (MG-N/L/HR)')
1254            WRITE(5,IMFT) (KO(I),(ANPOT(J,I),J=1,NR,1),I=1,NL,KH)
1255            WRITE(5,416)
1256          416 FORMAT(1H0,3X,'NO3    (MG-N/L)')
1257            WRITE(5,IMFT) (KO(I),(NO3(J,I),J=1,NR,1),I=1,NL,KH)
1258            WRITE(5,461)
1259          461 FORMAT(1H0,3X,'ALK    (MG-CACO3/L)')
1260            WRITE(5,IMFT) (KO(I),(ALK(J,I),J=1,NR,1),I=1,NL,KH)
1261            IF(RESULT.NE.0) GO TO 127
1262         126   WRITE(18,522)ECOD,ENT,ENH3,ENO3,EALK
1263            WRITE(18,522)KOM,D,INIT(1),ANOX(1),AROB(1),IFREQ,NDH
1264         522   FORMAT()
1265            WRITE(18,45)(OT(1,I),XV(1,I),ST(1,I),NT(1,I),NH3(1,I),NO3(1,I),
1266           &ALK(1,I),I=1,NDH)
1267            WRITE(18,522) START,END
1268         45    FORMAT(7E13.5)
1269            ENDFILE 18
1270         127   STOP
1271            END
```

```
C*************************************************************
      SUBROUTINE UAERO(D,DT,J,FR,FCA,FCS,KVT,KMPT,KSPT,KMST,KSST,
     1P,NR,YH,BHT,F,KRT,FOE,FN,FOS,FNS,NJ,DDT,UMT,YN,KNT,RDT,BNT,
     2FACT,LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,I)
C*************************************************************
C            THIS SUBROUTINE CALCULATES THE DYNAMIC RESPONSE OF A
C              REACTOR IN A SERIES UNDER AEROBIC CONDITIONS
C*************************************************************
      PARAMETER NE=3
      PARAMETER NEE=4
      PARAMETER ND=241
      PARAMETER NDD=242
      INTEGER D
      DOUBLE PRECISION DT,DDT,RDT
      COMMON/EXTC/SBS(NE,ND),SU(NE,ND),XA(NE,ND),XE(NE,ND),XI(NE,ND),
     1XV(NE,ND),XS(NE,ND),NU(NE,ND),NON(NE,ND),XN(NE,ND),NH3(NE,ND),
     2NO3(NE,ND),SBP(NE,ND),ALK(NE,ND)
      COMMON/BLOCKC/Q(NEE,NDD),V(NE),SBI(ND),SUI(ND),NIT(NE,ND),
     1NUI(ND),XII(ND),NONI(ND),NH3I(ND),NO3I(ND),SBPR(ND),ALKI(ND),
     2XAR(ND),XSR(ND),XER(ND),XIR(ND),XNR(ND),SYNTH(NE,ND),QAV(NEE),
     3ANPOT(NE,ND),AMON(NE,ND)
      REAL KVT,KMPT,KSPT,KMST,KSST,KRT,KNT
      REAL NUI,NONI,NH3I,NO3I,NIT
      REAL NU,NON,NH3,NO3
      REAL NH3T,NITT,NNIT,NO3T
C*************************************************************
      CALL UDUMV(J,I,DUMIF,DUMS,DUMA,DUMB,DUMIR,DUMO,
     1LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,
     1DT,DDT,RDT)
C*************************************************************
      K=J-1
      IF(K.EQ.0) K=1
      SCER=P*(1.0-F)*BHT*XA(J,I)*DDT
      SCERN=FN*SCER/P
      CVNON=KRT*XA(J,I)*NON(J,I)*DDT
      SBST=SBS(J,I)
      SBST=SBS(J,I)
      NH3T=NH3(J,I)
      NO3T=NO3(J,I)
      XNT=XN(J,I)
      SCVO=0.00
      SCIS=0.00
      NNIT=0.0
      ANPOTS=0.0
      DO 212 IJ=1,NJ,1
      SCIST=KMST*SBST/(KSST+SBST)*XA(J,I)*DDT
      SCVOT=KMPT*XS(J,I)/(KSPT*XA(J,I)+XS(J,I)*P)*XA(J,I)*DDT
      SYNTHT=SCVOT*P+SCIST
      NITT=(UMT/YN)*NH3T/(KNT+NH3T)*XNT*DDT
      SBST=SBST+DUMIF*FCA*SBI(I)-SCIST
     1+DUMS*SBS(NR,I)+DUMA*SBS(LARO,I)-DUMO*SBST
     1+DUMB*SBS(LBRO,I)+DUMIR*SBS(K,I)
      IF(SBST.LE.0.0) SBST=0.0
      TNH3T=NH3T+DUMIF*NH3I(I)+CVNON-NITT
     1+DUMS*NH3(NR,I)+DUMA*NH3(LARO,I)-DUMO*NH3T
     1+DUMB*NH3(LBRO,I)+DUMIR*NH3(K,I)
     1+(1.0-FOE)*SCERN-(1.0-FOS)*FN*YH*SYNTHT
      IF(TNH3T.GT.0.0) GO TO 433
      NH3T=0.0
      NO3T=NO3T+DUMIF*NO3I(I)+TNH3T
     1+DUMS*NO3(NR,I)+DUMA*NO3(LARO,I)-DUMO*NO3T
     1+DUMB*NO3(LBRO,I)+DUMIR*NO3(K,I)
      GO TO 444
```

```
 64        433 NH3T=TNH3T
 65            NO3T=NO3T+DUMIF*NO3I(I)+NITT
 66           1+DUMS*NO3(NR,I)+DUMA*NO3(LARO,I)-DUMO*NO3T
 67           1+DUMB*NO3(LBRO,I)+DUMIR*NO3(K,I)
 68       444  XNT=XNT+YN*NITT-BNT*XNT*DDT
 69           1+DUMS*XNR(I)+DUMA*XN(LARO,I)-DUMO*XNT
 70           1+DUMB*XN(LBRO,I)+DUMIR*XN(K,I)
 71            SCVO=SCVO+SCVOT
 72            SCIS=SCIS+SCIST
 73            NNIT=NNIT+NITT
 74        212 CONTINUE
 75            SYNTH(J,I)=SCVO*P+SCIS
 76            NIT(J,I)=NNIT
 77            ANPOT(J,I)=ANPOTS
 78            SBS(J,I+1)=SBST
 79            NH3(J,I+1)=NH3T
 80            NO3(J,I+1)=NO3T
 81            XN(J,I+1)=XNT
 82            DUMIF=DUMIF/RDT
 83            DUMIR=DUMIR/RDT
 84            DUMA=DUMA/RDT
 85            DUMB=DUMB/RDT
 86            DUMS=DUMS/RDT
 87            DUMO=DUMO/RDT
 88            BRAK=FR-(XS(J,I)/(XA(J,I)))
 89            IF(BRAK.GT.0.00) GO TO 40
 90            BRAK=0.00
 91            XS(J,I)=FR*(XA(J,I))
 92        40  XV(J,I)=XS(J,I)+XA(J,I)+XE(J,I)+XI(J,I)
 93            SCIP=DT*KVT*BRAK*SBP(J,I)*XA(J,I)
 94            SBP(J,I+1)=SBP(J,I)+DUMIF*(1.0-FCA)*SBI(I)-SCIP
 95           1+DUMS*SBPR(I)+DUMA*SBP(LARO,I)-DUMO*SBP(J,I)
 96           1+DUMB*SBP(LBRO,I)+DUMIR*SBP(K,I)
 97           2+(1.0-FCS)*SCER/RDT
 98            IF(SBP(J,I+1).LT.0.0) SBP(J,I+1)=0.0
 99            XS(J,I+1)=XS(J,I)+SCIP/P-SCVO
100           1+DUMS*XSR(I)+DUMA*XS(LARO,I)-DUMO*XS(J,I)
101           1+DUMB*XS(LBRO,I)+DUMIR*XS(K,I)
102            IF(XS(J,I+1).LT.0.0) XS(J,I+1)=0.0
103            SU(J,I+1)=SU(J,I)+DUMIF*SUI(I)
104           1+DUMS*SU(NR,I)+DUMA*SU(LARO,I)-DUMO*SU(J,I)
105           1+DUMB*SU(LBRO,I)+DUMIR*SU(K,I)
106            NU(J,I+1)=NU(J,I)+DUMIF*NUI(I)
107           1+DUMS*NU(NR,I)+DUMA*NU(LARO,I)-DUMO*NU(J,I)
108           1+DUMB*NU(LBRO,I)+DUMIR*NU(K,I)
109            XA(J,I+1)=XA(J,I)+YH*SYNTH(J,I)-BHT*XA(J,I)*DT
110           1+DUMS*XAR(I)+DUMA*XA(LARO,I)-DUMO*XA(J,I)
111           1+DUMB*XA(LBRO,I)+DUMIR*XA(K,I)
112            XE(J,I+1)=XE(J,I)+F*BHT*XA(J,I)*DT
113           1+DUMS*XER(I)+DUMA*XE(LARO,I)-DUMO*XE(J,I)
114           1+DUMB*XE(LBRO,I)+DUMIR*XE(K,I)
115            XI(J,I+1)=XI(J,I)+DUMIF*XII(I)
116           1+DUMS*XIR(I)+DUMA*XI(LARO,I)-DUMO*XI(J,I)
117           1+DUMB*XI(LBRO,I)+DUMIR*XI(K,I)
118            NON(J,I+1)=NON(J,I)+DUMIF*NONI(I)-CVNON/RDT
119           1+DUMS*NON(NR,I)+DUMA*NON(LARO,I)-DUMO*NON(J,I)
120           1+DUMB*NON(LBRO,I)+DUMIR*NON(K,I)
121           2+FOE*SCERN/RDT-FOS*FN*YH*SYNTH(J,I)
122           3-FNS*(SCIP/P-SCVO)
123            AMON(J,I)=(CVNON/RDT+(1.0-FOE)*SCERN/RDT)
124           1-(1.0-FOS)*FN*YH*SYNTH(J,I)
125            ALK(J,I+1)=ALK(J,I)+DUMIF*ALKI(I)
126           1+DUMS*ALK(NR,I)+DUMA*ALK(LARO,I)-DUMO*ALK(J,I)
127           1+DUMB*ALK(LBRO,I)+DUMIR*ALK(K,I)
```

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128           1-7.14*NIT(J,I)+3.57*AMON(J,I)
129            XV(J,I+1)=XS(J,I+1)+XA(J,I+1)+XE(J,I+1)+XI(J,I+1)
130            RETURN
131            END
```

```
C*************************************************************
      SUBROUTINE UANOX(D,DT,J,FR,FCA,FCS,KVT,KMPT,KSPT,KMST,KSST,
     1P,NR,YH,BHT,F,KRT,FOE,FN,FOS,FNS,NJ,DDT,UMT,YN,KNT,RDT,BNT,
     2FACT,LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,I)
C*************************************************************
C            THIS SUBROUTINE CALCULATES THE DYNAMIC RESPONSE OF A
C              REACTOR IN A SERIES UNDER ANOXIC CONDITIONS
C*************************************************************
      PARAMETER NE=3
      PARAMETER NEE=4
      PARAMETER ND=241
      PARAMETER NDD=242
      INTEGER D
      DOUBLE PRECISION DT,DDT,RDT
      COMMON/EXTC/SBS(NE,ND),SU(NE,ND),XA(NE,ND),XE(NE,ND),XI(NE,ND),
     1XV(NE,ND),XS(NE,ND),NU(NE,ND),NON(NE,ND),XN(NE,ND),NH3(NE,ND),
     2NO3(NE,ND),SBP(NE,ND),ALK(NE,ND)
      COMMON/BLOCKC/Q(NEE,NDD),V(NE),SBI(ND),SUI(ND),NIT(NE,ND),
     1NUI(ND),XII(ND),NONI(ND),NH3I(ND),NO3I(ND),SBPR(ND),ALKI(ND),
     2XAR(ND),XSR(ND),XER(ND),XIR(ND),XNR(ND),SYNTH(NE,ND),QAV(NEE),
     3ANPOT(NE,ND),AMON(NE,ND)
      REAL KVT,KMPT,KSPT,KMST,KSST,KRT,KNT
      REAL NUI,NONI,NH3I,NO3I,NIT
      REAL NU,NON,NH3,NO3
      REAL NH3T,NITT,NNIT,NO3T
      RNCOD=2.85/(1.0-P*YH)
C*************************************************************
      CALL UDUMV(J,I,DUMIF,DUMS,DUMA,DUMB,DUMIR,DUMO,
     1LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,
     1DT,DDT,RDT)
C*************************************************************
      K=J-1
      IF(K.EQ.0) K=1
      SCER=P*(1.0-F)*BHT*XA(J,I)*DDT
      SCERN=FN*SCER/P
      CVNON=KRT*FACT*XA(J,I)*NON(J,I)*DDT
      SCVOD=KMPT*XS(J,I)/(KSPT*XA(J,I)+XS(J,I)*P)*FACT*XA(J,I)*DDT
      SCVO=0.00
      SCIS=0.00
      NNIT=0.00
      ANPOTS=0.00
      SBST=SBS(J,I)
      SBST=SBS(J,I)
      NH3T=NH3(J,I)
      NO3T=NO3(J,I)
      XNT=XN(J,I)
      DO 213 IJ=1,NJ,1
      SCIST=KMST*SBST/(KSST+SBST)*FACT*XA(J,I)*DDT
      SCVOT=KMPT*XS(J,I)/(KSPT*XA(J,I)+XS(J,I)*P)*FACT*XA(J,I)*DDT
      SYNTHT=SCVOT*P+SCIST
      DENCAP=SYNTHT/RNCOD
      DENIT=DENCAP
      RDEN=SCIST/SYNTHT
      NITT=0.0
      ANPOTT=0.00
      IF((NO3T-DENIT).GT.0.00) GO TO 13
      DENIT=NO3T
      SYNTHT=DENIT*RNCOD
      SCIST=SYNTHT*RDEN
      SCVOT=SYNTHT*(1.0-RDEN)/P
      ANPOTT=DENCAP-DENIT
   13 CONTINUE
      SCVO=SCVO+SCVOT
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```
      SCIS=SCIS+SCIST
      ANPOTS=ANPOTS+ANPOTT
      SBST=SBST+DUMIF*FCA*SBI(I)-SCIST
     1+DUMS*SBS(NR,I)+DUMA*SBS(LARO,I)-DUMO*SBST
     1+DUMB*SBS(LBRO,I)+DUMIR*SBS(K,I)
      IF(SBST.LE.0.0) SBST=0.0
      NITT=0.00
      TNH3T=NH3T+DUMIF*NH3I(I)+CVNON
     1+DUMS*NH3(NR,I)+DUMA*NH3(LARO,I)-DUMO*NH3T
     1+DUMB*NH3(LBRO,I)+DUMIR*NH3(K,I)
     1+(1.0-FOE)*SCERN-(1.0-FOS)*FN*YH*SYNTHT
      IF(TNH3T.GT.0.0) GO TO 43
      NH3T=0.0
      NO3T=NO3T+DUMIF*NO3I(I)+NITT-DENIT+TNH3T
     1+DUMS*NO3(NR,I)+DUMA*NO3(LARO,I)-DUMO*NO3T
     1+DUMB*NO3(LBRO,I)+DUMIR*NO3(K,I)
      IF(NO3T.LT.0.00) NO3T=0.00
      GO TO 44
   43 NH3T=TNH3T
      NO3T=NO3T+DUMIF*NO3I(I)+NITT-DENIT
     1+DUMS*NO3(NR,I)+DUMA*NO3(LARO,I)-DUMO*NO3T
     1+DUMB*NO3(LBRO,I)+DUMIR*NO3(K,I)
   44 XNT=XNT+YN*NITT-BNT*XNT*DDT
     1+DUMS*XNR(I)+DUMA*XN(LARO,I)-DUMO*XNT
     1+DUMB*XN(LBRO,I)+DUMIR*XN(K,I)
  213 CONTINUE
      SYNTH(J,I)=SCVO*P+SCIS
      NIT(J,I)=NNIT
      ANPOT(J,I)=ANPOTS
      SBS(J,I+1)=SBST
      NH3(J,I+1)=NH3T
      NO3(J,I+1)=NO3T
      XN(J,I+1)=XNT
      DUMIF=DUMIF/RDT
      DUMIR=DUMIR/RDT
      DUMA=DUMA/RDT
      DUMB=DUMB/RDT
      DUMS=DUMS/RDT
      DUMO=DUMO/RDT
      BRAK=FR-(XS(J,I)/(XA(J,I)))
      IF(BRAK.GT.0.00) GO TO 40
      BRAK=0.00
      XS(J,I)=FR*(XA(J,I))
   40 XV(J,I)=XS(J,I)+XA(J,I)+XE(J,I)+XI(J,I)
      SCIP=DT*KVT*BRAK*SBP(J,I)*XA(J,I)
      SBP(J,I+1)=SBP(J,I)+DUMIF*(1.0-FCA)*SBI(I)-SCIP
     1+DUMS*SBPR(I)+DUMA*SBP(LARO,I)-DUMO*SBP(J,I)
     1+DUMB*SBP(LBRO,I)+DUMIR*SBP(K,I)
     2+(1.0-FCS)*SCER/RDT
      IF(SBP(J,I+1).LT.0.0) SBP(J,I+1)=0.0
      XS(J,I+1)=XS(J,I)+SCIP/P-SCVO
     1+DUMS*XSR(I)+DUMA*XS(LARO,I)-DUMO*XS(J,I)
     1+DUMB*XS(LBRO,I)+DUMIR*XS(K,I)
      IF(XS(J,I+1).LT.0.0) XS(J,I+1)=0.0
      SU(J,I+1)=SU(J,I)+DUMIF*SUI(I)
     1+DUMS*SU(NR,I)+DUMA*SU(LARO,I)-DUMO*SU(J,I)
     1+DUMB*SU(LBRO,I)+DUMIR*SU(K,I)
      NU(J,I+1)=NU(J,I)+DUMIF*NUI(I)
     1+DUMS*NU(NR,I)+DUMA*NU(LARO,I)-DUMO*NU(J,I)
     1+DUMB*NU(LBRO,I)+DUMIR*NU(K,I)
      XA(J,I+1)=XA(J,I)+YH*SYNTH(J,I)-BHT*XA(J,I)*DT
     1+DUMS*XAR(I)+DUMA*XA(LARO,I)-DUMO*XA(J,I)
     1+DUMB*XA(LBRO,I)+DUMIR*XA(K,I)
      XE(J,I+1)=XE(J,I)+F*BHT*XA(J,I)*DT
```

```
128           1+DUMS*XER(I)+DUMA*XE(LARO,I)-DUMO*XE(J,I)
129           1+DUMB*XE(LBRO,I)+DUMIR*XE(K,I)
130            XI(J,I+1)=XI(J,I)+DUMIF*XII(I)
131           1+DUMS*XIR(I)+DUMA*XI(LARO,I)-DUMO*XI(J,I)
132           1+DUMB*XI(LBRO,I)+DUMIR*XI(K,I)
133            NON(J,I+1)=NON(J,I)+DUMIF*NONI(I)-CVNON/RDT
134           1+DUMS*NON(NR,I)+DUMA*NON(LARO,I)-DUMO*NON(J,I)
135           1+DUMB*NON(LBRO,I)+DUMIR*NON(K,I)
136           2+FOE*SCERN/RDT-FOS*FN*YH*SYNTH(J,I)
137           3-FNS*(SCIP/P-SCVO)
138            AMON(J,I)=(CVNON/RDT+(1.0-FOE)*SCERN/RDT)
139           1-(1.0-FOS)*FN*YH*SYNTH(J,I)
140            ALK(J,I+1)=ALK(J,I)+DUMIF*ALKI(I)
141           1+DUMS*ALK(NR,I)+DUMA*ALK(LARO,I)-DUMO*ALK(J,I)
142           1+DUMB*ALK(LBRO,I)+DUMIR*ALK(K,I)
143           1+3.57*(SYNTH(J,I)/RNCOD+AMON(J,I))
144            XV(J,I+1)=XS(J,I+1)+XA(J,I+1)+XE(J,I+1)+XI(J,I+1)
145            RETURN
146            END
```

```
      C*****************************************************************
            SUBROUTINE USETL(NL,J,QSR,I)
      C*****************************************************************
      C             THIS SUBROUTINE CALCULATES THE DYNAMIC RESPONSE OF THE
      C              SETTLING TANK
      C*****************************************************************
            PARAMETER NE=3
            PARAMETER NEE=4
            PARAMETER ND=241
            PARAMETER NDD=242
            COMMON/EXTC/SBS(NE,ND),SU(NE,ND),XA(NE,ND),XE(NE,ND),XI(NE,ND),
           1XV(NE,ND),XS(NE,ND),NU(NE,ND),NON(NE,ND),XN(NE,ND),NH3(NE,ND),
           2NO3(NE,ND),SBP(NE,ND),ALK(NE,ND)
            COMMON/BLOCKC/Q(NEE,NDD),V(NE),SBI(ND),SUI(ND),NIT(NE,ND),
           1NUI(ND),XII(ND),NONI(ND),NH3I(ND),NO3I(ND),SBPR(ND),ALKI(ND),
           2XAR(ND),XSR(ND),XER(ND),XIR(ND),XNR(ND),SYNTH(NE,ND),QAV(NEE),
           3ANPOT(NE,ND),AMON(NE,ND)
            REAL NUI,NONI,NH3I,NO3I,NIT
            REAL NU,NON,NH3,NO3
            DUM=(QSR+Q(J+1,I))/QSR
            XAR(I+1)=XA(J,I+1)*DUM
            XSR(I+1)=XS(J,I+1)*DUM
            XER(I+1)=XE(J,I+1)*DUM
            XIR(I+1)=XI(J,I+1)*DUM
            SBPR(I+1)=SBP(J,I+1)*DUM
            XNR(I+1)=XN(J,I+1)*DUM
            RETURN
            END
```

```
      C*****************************************************************
            SUBROUTINE UDUMV(J,I,DUMIF,DUMS,DUMA,DUMB,DUMIR,DUMO,
           1LINI,LSRI,LARI,LARO,LBRI,LBRO,QSR,QAR,QBR,
           1DT,DDT,RDT)
      C*****************************************************************
      C              THIS SUBROUTINE CALCULATES THE DUMMY VALUES
      C              FROM THE PROCESS CONFIGURATION INPUT DATA
      C*****************************************************************
            PARAMETER NE=3
            PARAMETER NEE=4
            PARAMETER ND=241
            PARAMETER NDD=242
            INTEGER D
            DOUBLE PRECISION DT,DDT,RDT
            COMMON/EXTC/SBS(NE,ND),SU(NE,ND),XA(NE,ND),XE(NE,ND),XI(NE,ND),
           1XV(NE,ND),XS(NE,ND),NU(NE,ND),NON(NE,ND),XN(NE,ND),NH3(NE,ND),
           2NO3(NE,ND),SBP(NE,ND),ALK(NE,ND)
            COMMON/BLOCKC/Q(NEE,NDD),V(NE),SBI(ND),SUI(ND),NIT(NE,ND),
           1NUI(ND),XII(ND),NONI(ND),NH3I(ND),NO3I(ND),SBPR(ND),ALKI(ND),
           2XAR(ND),XSR(ND),XER(ND),XIR(ND),XNR(ND),SYNTH(NE,ND),QAV(NEE),
           3ANPOT(NE,ND),AMON(NE,ND)
            DUMIF=0.0
            IF(J.EQ.LINI) DUMIF=DDT*Q(J,I)/V(J)
            DUMS=0.0
            IF(J.EQ.LSRI) DUMS=DDT*QSR/V(J)
            DUMA=0.0
            IF(J.EQ.LARI.AND.J.LT.LARO) DUMA=DDT*QAR/V(J)
            DUMB=0.0
            IF(J.EQ.LBRI.AND.J.LT.LBRO) DUMB=DDT*QBR/V(J)
            IF(J.EQ.1) GO TO 11
            IF(J.LE.LINI.AND.J.GT.LSRI) DUMIR=DDT*QSR/V(J)
            IF(J.LE.LINI.AND.J.LE.LSRI.AND.J.GT.LBRI.AND.J.LE.LBRO)
           1DUMIR=DDT*QBR/V(J)
            IF(J.LE.LINI.AND.J.LE.LSRI.AND.J.GT.LARI.AND.J.LE.LARO)
           1DUMIR=DDT*QAR/V(J)
            IF(J.LE.LINI.AND.J.LE.LSRI.AND.J.GT.LARI.AND.J.LE.LARO.
           1AND.J.GT.LBRI.AND.J.LE.LBRO) DUMIR=DDT*(QAR+QBR)/V(J)
            IF(J.LE.LINI.AND.J.GT.LSRI.AND.J.GT.LARI.AND.J.LE.LARO)
           1DUMIR=DDT*(QSR+QAR)/V(J)
            IF(J.LE.LINI.AND.J.GT.LSRI.AND.J.GT.LBRI.AND.J.LE.LBRO)
           1DUMIR=DDT*(QSR+QBR)/V(J)
            IF(J.LE.LINI.AND.J.GT.LSRI.AND.J.GT.LARI.AND.J.LE.LARO.
           1AND.J.GT.LBRI.AND.J.LE.LBRO) DUMIR=DDT*(QSR+QAR+QBR)/V(J)
            IF(J.GT.LINI.AND.J.GT.LSRI) DUMIR=DDT*(Q(J,I)+QSR)/V(J)
            IF(J.GT.LINI.AND.J.LE.LSRI.AND.J.GT.LBRI.AND.J.LE.LBRO)
           1DUMIR=DDT*(Q(J,I)+QBR)/V(J)
            IF(J.GT.LINI.AND.J.LE.LSRI.AND.J.GT.LARI.AND.J.LE.LARO)
           1DUMIR=DDT*(Q(J,I)+QAR)/V(J)
            IF(J.GT.LINI.AND.J.GT.LSRI.AND.J.GT.LARI.AND.J.LE.LARO)
           1DUMIR=DDT*(Q(J,I)+QAR+QSR)/V(J)
            IF(J.GT.LINI.AND.J.GT.LSRI.AND.J.GT.LBRI.AND.J.LE.LBRO)
           1DUMIR=DDT*(Q(J,I)+QBR+QSR)/V(J)
            IF(J.GT.LINI.AND.J.GT.LSRI.AND.J.GT.LARI.AND.J.LE.LARO.
           1AND.J.GT.LBRI.AND.J.LE.LBRO) DUMIR=DDT*(Q(J,I)+QSR+QAR+QBR)/V(J)
         11 CONTINUE
            IF(J.EQ.1) DUMIR=0.0
            DUMO=DUMIF+DUMIR+DUMS+DUMA+DUMB
            RETURN
            END
```

APPENDIX A3

RESULTS OF THE BENCH SCALE EXPERIMENTS

Experimental response of a bench scale lab unit operated by Ekama (1976) and consisting of an anoxic reactor (2,5ℓ) followed by two aerobic reactors with volumes of 2,5 and 7,5ℓ respectively. Two series of tests were carried out. For operational conditions see Table 4.3 and Fig 4.4 in chapter 4. The following data was recorded during the two series of tests :

Table A3.1a and A3.1b: Oxygen uptake rates in the aerobic reactors, average oxygen uptake rate and mean temperature.

Table A3.2a and A3.2b: COD concentration in the influent, filtered mixed liquor in the reactor and in the effluent.

Table A3.3a and A3.3b: VSS concentration in the reactor and weighted average VSS.

Table A3.4a and A3.4b: TKN concentrations in the influent, filtered mixed liquor in the reactors and in the effluent.

Table A3.5a and A3.5b: Ammonia concentration in filtered mixed liquor of the reactors and in the effluent.

Table A3.6a and A3.6b: Nitrate concentration in filtered mixed liquor of the reactors and in the effluent.

In the tables the numbers 1, 2 and 3 refer to the first (anoxic), second and third reactor respectively. Inf refers to influent and E refers to effluent.

Oxygen uptake rate in mg $O.\ell^{-1}h^{-1}$
Temperature in °C
COD concentration in mg $COD.\ell^{-1}$
Volatile solids concentration in mg $VSS.\ell^{-1}$
TKN concentration in mg $N.\ell^{-1}$
Ammonia concentration in mg $NH_3\text{-}N.\ell^{-1}$
Nitrate concentration in mg $NO_3\text{-}N.\ell^{-1}$

Table A3.1a Oxygen uptake rates, average oxygen uptake rate and temperature during the first series of tests.

DATE	2	3	$\overline{X}$	Temp
14.10	65,2	22,7	33,4	26
15.10	79,3	29,1	41,7	25
16.10	64,0	25,6	35,2	26
17.10	57,2	14,6	25,1	20
18.10	56,5	19,3	28,6	20
19.10	83,2	17,7	34,1	25
20.10	94,8	26,1	43,3	26
21.10	106,4	25,4	45,7	27
22.10	103,1	20,5	41,1	27
23.10	-	-	-	-
24.10	105,8	20,3	41,6	27
25.10	106,5	21,9	41,5	26
26.10	106,0	20,3	41,7	27
Average	85,2	21,9	37,7	25,1

Table A3 .2a : COD concentrations in influent, filtered mixed liquor in the reactors and in the effluent, during the first series of tests.

DATE	Inf	1	2	3	E
14.10.76					16,5
15.10.76	454	34,0	33,1	27,6	19,7
16.10.76	637				17,3
17.10.76	536	22,7	20,2	22,7	17,3
18.10.76	471				15,0
19.10.76	364	26,4	19,5	19,6	17,4
20.10.76	537				23,7
21.10.76	556				
22.10.76	477	30,3	75,1	24,3	17,5
23.10.76	466				13,6
24.10.76	440	26,4	20,5	18,8	19,4
25.10.76	442				15,3
26.10.76	512	23,7	18,8	21,2	18,5
Average	490	27,3	22,9	22,4	

For symbols and units see page A3.1

Table A3.3a : VSS concentrations in the reactor and weighted average during the first series of tests.

DATE	Inf	1	2	3	$\overline{X}$
14.10.76		2166	2108	2137	
15.10.76					
16.10.76		1956	2062	1966	1995
17.10.76					
18.10.76		2198	2134	2108	2143
19.10.76					
20.10.76		2146	2180	1984	2103
21.10.76					
22.10.76		2184	2320	2176	2260
23.10.76					
24.10.76		2270	2156	1964	2130
25.10.76					
26.10.76		2172	2210	2186	2189
Average		2170	2177	2064	2137

For symbols and units see page A3.1

Table A3.4a *TKN concentrations in the influent, filtered mixed liquor TKN in the reactors and in the effluent during the first series of tests.*

DATE	Inf	1	2	3	E
14.10.76	41,1	9,6	5,3	2,1	1,4
15.10.76	61,6	11,7	8,3	2,1	1,8
16.10.76	45,2	8,0	3,3	1,6	2,4
17.10.76	41,5	12,7	6,2	3,9	5,1
18.10.76	47,6	10,1	5,8	0,9	1,8
19.10.76	45,1	8,6	3,4	2,2	0,0
20.10.76	45,1	10,4	4,6	1,7	1,4
21.10.76	39,3	11,1	3,9	0,9	1,8
22.10.76	48,0	9,2	6,9	0,9	1,4
23.10.76	39,2	-	-	-	1,1
24.10.76	32,8	8,0	1,4	1,9	2,4
25.10.76	50,0	16,4	6,9	0,0	1,4
26.10.76	50,8	9,8	6,8	2,1	2,2
Average	47,6	10,0	5,5	1,7	2,0

For symbols and units see page A3.1

Table A3.5a : Concentrations of ammonia in the reactors and in the effluent during the first series of tests.

DATE	Inf	1	2	3	E
14.10.76		9,1	4,1	0,0	6,0
15.10.76		12,8	8,8	0,0	0,0
16.10.76		8,5	4,0	0,0	0,0
17.10.76		7,4	3,0	0,0	0,0
18.10.76		9,2	5,4	0,0	0,0
19.10.76		6,9	0,0	0,0	0,0
20.10.76		11,1	5,6	0,0	0,0
21.10.76		9,5	3,1	0,0	0,0
22.10.76		12,0	7,1	0,0	0,0
23.10.76		-	-	-	0,0
24.10.76		9,4	2,3	0,0	0,0
25.10.76		10,7	4,1	0,0	0,0
26.10.76		11,0	2,6	0,0	0,0
Average		9,8	4,2	0,0	0,0

For symbols and units see page A3.1

Table A3 .6a:Nitrate and nitrite concentration in the reactors and in the effluent during the first series of tests.

DATE	Inf	1	2	3	E
14.10.76		1,2	5,5	10,6	10,2
15.10.76		0,4	3,6	10,9	10,9
16.10.76		2,0	6,0	10,8	8,5
17.10.76		1,0	4,9	8,0	7,9
18.10.76		5,1	8,1	14,8	12,8
19.10.76		2,4	7,5	11,9	10,9
20.10.76		0,4	5,0	10,0	9,1
21.10.76		0,6	6,0	8,4	8,2
22.10.76		0,6	4,0	8,8	7,9
23.10.76		-	-	-	6,0
24.10.76		0,5	5,8	8,0	7,5
25.10.76		0,4	4,9	9,0	8,4
26.10.76		6,4	5,8	7,8	7,4
Average		1,2	5,6	9,8	9,2

For symbols and units see page A3.1

Table A3.1b Oxygen uptake rates, average oxygen uptake rate and temperature during the second series of tests.

DATE	2	3	$\overline{X}$	Temp
27.10	-	-	-	-
28.10	114,6	25,1	47,5	28
29.10	85,7	12,7	30,9	26
30.10	85,7	15,3	33,3	27
31.10	102,6	16,2	37,8	27
1.11	84,6	14,8	32,3	26
2,11	77,9	13,8	29,8	20
3.11	73,2	12,6	27,8	21
4.11	69,8	13,0	27,2	18
5.11	89,0	17,0	35,0	22
6.11	-	-	-	-
7.11	77,5	17,4	32,4	24
8.11	105,9	21,4	42,5	26
9.11	78,9	12,7	29,2	21
Average	87,1	16,0	33,8	23,8

Table A3.2b COD concentrations in the influent, filtered mixed liquor in the reactors and in the effluent during the second series of tests.

DATE	Inf	1	2	3	E
27.10	515	28,2	19,7	16,1	17,9
28.10	483				17,9
29.10	460	24,9	17,0	20,0	14,9
30.10	522				20,3
31.10	476	18,7	16,5	18,7	16,3
1.11	533				15,6
2.11	447	33,3	26,6	23,4	15,5
3.11	410				17,3
4.11	462	37,5	30,0	22,4	14,0
5.11	376				13,5
6.11	462	46,6	20,4	13,7	14,3
7.11	451				9,3
8.11	360	21,0	15,7	14,8	15,2
9.11	453				15,4
Average	461	30,0	20,8	18,4	15,4

Table A3.3b *VSS concentration in the reactor and weighted average during the second series of tests.*

DATE	Inf	1	2	3	$\overline{X}$
27.10.76					
28.10.76		2322	2352	2296	2333
29.10.76					
30.10.76		2400	2398	2296	2365
31.10.76					
1.11.76		2520	2446	2434	2467
2.11.76					
3.11.76		2376	2382	2406	2388
4.11.76					
5.11.76		2676	2510	2498	2561
6.11.76					
7.11.76		2507	2473	2360	2447
8.11.76					
9.11.76		2650	2677	2410	2560
Average		2493	2546	2386	2444

For symbols and units see page A3.1

$\overline{X}$ = weighted average VSS

Table A3.4b TKN concentrations in the influent, filtered mixed liquor TKN in the reactors and in the effluent during the second series of tests.

DATE	Inf	1	2	3	E
27.10.76	47,3	-	-	-	1,2
28.10.76	46,4	8,0	6,4	1,9	2,8
29.10.76	49,2	7,6	4,1	0,9	2,3
30.10.76	55,6	7,3	1,8	1,7	2,8
31.10.76	48,2	10,5	6,6	5,5	3,3
1.11.76	51,5	9,4	4,8	1,7	2,4
2.11.76	55,0	12,4	11,7	4,5	4,3
3.11.76	50,1	7,6	5,7	2,1	2,4
4.11.76	42,2	0,5	2,5	1,1	1,9
5.11.76	32,9	6,6	3,0	2,1	2,2
6.11.76	37,8	5,1	2,4	1,5	2,1
7.11.76	44,3	4,0	2,2	0,9	1,4
8.11.76	37,5	8,6	6,1	0,9	2,8
9.11.76	43,7	10,5	6,1	1,1	3,7
Average	46,1	7,6	4,3	1,7	2,5

For symbols and units see page A3.1

Table A3.5b Concentration of ammonia in the reactors and in the effluent during the second series of tests.

DATE	Inf	1	2	3	E
27.10.76		-	-	-	0,0
28.10.76		8,2	4,8	0,0	0,0
29.10.76		5,0	6,0	0,0	0,0
30.10.76		6,2	0,0	0,0	0,0
31.10.76		7,2	2,9	4,0	0,0
1.11.76		8,0	2,2	0,0	0,0
2.11.76		> 16	> 16	10,g	2,4
3.11.76		3,5	0,0	0,0	0,0
4.11.76		3,7	0,0	0,0	0,0
5.11.76		2,8	6,0	0,0	0,0
6.11.76		2,8	0,0	0,0	0,0
7.11.76		4,0	0,0	0,0	0,0
8.11.76		3,9	0,0	0,0	0,0
9.11.76		4,8	0,0	0,0	0,0
Average		5,0	0,7	0,1	0,0

For symbols and units see page A3.1

Table A3.6b *Nitrate & nitrite concentrations in the reactors and in the effluent during the second series of tests.*

DATE	Inf	1	2	3	E
27.10.76		-	-	-	5,9
28.10.76		6,4	3,1	5,7	5,7
29.10.76		1,6	5,4	7,0	6,3
30.10.76		3,1	7,9	9,5	6,7
31.10.76		0,5	3,9	8,8	5,6
1.11.76		5,6	9,2	9,2	10,1
2.11.76		0,4	0,6	3,9	9,4
3.11.76		6,3	10,7	11,7	12,0
4.11.76		8,0	12,6	13,4	13,2
5.11.76		3,7	9,6	10,8	10,6
6.11.76		3,4	7,0	9,8	9,6
7.11.76		3,2	9,6	10,0	9,6
8.11.76		5,4	8,0	10,3	10,0
9.11.76		3,2	7,6	9,2	9,5
Average		3,3	7,2	8,9	8,9

For symbols and units see page A3.1

APPENDIX A4

RESULTS OF THE PILOT SCALE EXPERIMENTS

Experimental response of a pilot scale unit operated by Ekama and Marais (1976) and consisting of a series of 5 equal sized reactors of 5 m^3 each, the first of which was anoxic and the rest aerobic. Two series of tests were carried out. For operational conditions see Table (4.4) and Fig (4.6) in chapter 4. The following data were recorded during the two series of tests.

Tables A4.1a and A4.1b : Oxygen uptake rates in the aerobic reactors, average oxygen uptake rate and mean temperature.

Tables A4.2a and A4.2b : COD concentrations in the influent, filtered mixed liquor of the reactors and in the effluent.

Tables A4.3a and A4.3b : VSS concentrations in the reactors and weighted average VSS.

Tables A4.4a and A4.4b : TKN concentrations in the influent, filtered mixed liquor in the reactors and in the effluent.

Tables A4.5a and A4.5b : Ammonia concentrations in the filtered mixed liquor of the reactors and in the effluent.

Tables A4.6a and A4.6b : (Nitrate + Nitrite) concentrations in the filtered mixed liquor of the reactors and in the effluent.

In the Tables the numbers I, II, III, IV and V refer to the reactor number of the series reactor configuration. (I is an anoxic reactor, the rest are aerobic). Inf. refers to the influent concentration and E to the effluent concentration.

- Oxygen uptake rate in mg $O.\ell^{-1}h^{-1}$
- Temperature in °C
- COD concentration in mg $COD.\ell^{-1}$
- Volatile solids concentration in mg $VSS.\ell^{-1}$
- TKN concentration in mg $N.\ell^{-1}$
- Ammonia concentration in mg $NH_3.N.\ell^{-1}$
- Nitrate concentration in mg NO_3-$N.\ell^{-1}$

Table A4.1a *Oxygen uptake rates in the aerobic reactors, average oxygen uptake rate and temperature during the first series of tests.*

DATE	I	II	III	IV	V	X	T
5.10.76							
6.10.76		57,1	19,7	18,4	16,4	27,9	21,5
7.10.76		60,9	28,0	20,7	15,5	31,3	21,5
8.10.76		64,6	26,5	19,8	17,3	32,1	22,0
9.10.76		67,1	56,9	32,4	28,4	47,9	22,0
10.10.76		56,6	40,9	26,8	21,4	36,4	21,5
1.11.76		57,4	33,3	25,9	17,0	33,4	21,0
12.10.76		64,6	41,7	21,2	17,4	36,2	22,0
13.10.76		62,1	15,9	19,8	16,6	31,1	22,0
14.10.76		56,1	41,8	22,5	21,2	35,4	22,5
15.10.76		70,4	43,1	24,7	21,0	39,8	22,5
16.10.76		65,4	36,5	20,8	19,7	35,6	22,5
17.10.76		-	-	-	-	-	-
18.10.76		43,6	37,2	20,3	18,1	29,8	19,5
19.10.76		41,5	37,3	20,4	14,1	28,3	19,5
20.10.76		62,0	42,7	27,8	26,2	29,7	22,0
Average		59,3	36,5	23,5	19,3	34,6	21,6

For symbols and units see Page A4.1

X = weighted average oxygen uptake rate

T = temperature (°C)

Table A4.2a COD concentration of the influent, filtered mixed liquor in the reactor and effluent during the first series of tests.

DATE	Inf	I	II	III	IV	V	E
5.10.76	465						13,2
6.10.76	457	18,4	15,9	11,7	17,1	8,8	12,9
7.10.76	478						14,2
8.10.76	441	25,8	17,3	17,7	21,1	21,1	14,3
9.10.76	516						20,5
10.10.76	433	24,8	19,6	19,6	20,5	15,8	17,5
11.10.76	345						18,3
12.10.76	484	21,5	17,5	16,6	16,2	18,9	16,6
13.10.76	463						19,7
14.10.76	454	24,0	21,3	19,6	19,1	20,9	12,6
15.10.76	637						14,6
16.10.76	536	25,2	19,2	18,7	18,7	19,2	15,3
17.10.76	471						16,8
18.10.76	364	26,6	17,3	15,3	13,3	16,8	16,3
19.10.76	537						12,6
20.10.76	556	32,8	24,0	20,6	19,6	18,6	15,4
Average	477	24,9	19,0	17,5	18,2	17,5	15,7

For symbols and units see Page A4.1

Table A4.3a *VSS concentrations in the reactors and average VSS concentration during the first series of tests.*

DATE	Inf	I	II	III	IV	V	E
4.10.76 5.10.76		2650	2537	2630	2589	2381	2617
6.10.76		2752	2474	2523	2637	2728	2623
7.10.76							
8.10.76		2788	2808	2474	2594	2616	2656
9.10.76							
10.10.76		2286	]]]8	]8]0	2606	2610	2510
11.10.76							
12.10.76		2544	2574	2520	1990	2502	2426
13.10.76							
14.10.76		2562	2316	2274	2280	2298	2334
15.10.76							
16.10.76		2328	2278	2226	2274	2220	2265
17.10.76							
18.10.76		2498	2382	2374	2344	2376	2395
19.10.76							
20.10.76		2702	2518	2510	2524	2468	2546
Average		2550	2447	2466	2406	2477	2469

For symbols and units see Page A4.1

Table A4.4a *TKN concentration in the influent, filtered mixed liquor of the reactors and effluent during the first series of tests.*

DATE	Inf	I	II	III	IV	V	E
5.10.76	39,7	-	-	-	-	-	1,1
6.10.76	41,1	9,1	2,0	1,5	0,7	0,74	2,2
7.10.76	43,0	9,3	2,2	2,2	1,9	1,7	2,5
8.10.76	42,6	5,3	4,6	2,4	1,8	1,3	1,6
9.10.76	51,0	12,8	7,6	4,8	3,0	1,7	2,6
10.10.76	46,1	9,0	8,3	5,6	3,6	2,7	2,2
1.1 .76	39,0	7,8	3,0	3,0	2,5	2,2	2,1
12.10.76	46,7	11,8	3,0	2,0	1,8	1,8	1,6
13.10.76	45,5	9,0	3,1	3,0	1,7	1,5	2,4
14.10.76	41,1	5,1	4,0	2,2	3,5	1,8	3,6
15.10.76	61,6	11,0	6,5	4,4	2,2	1,5	2,0
16.10.76	45,2	8,0	3,3	1,9	1,9	2,1	1,2
17.10.76	41,5	-	-	-	-	-	2,1
18.10.76	47,6	15,4	9,7	6,6	5,7	3,9	1,8
19.10.76	45,1	8,9	5,3	3,3	1,8	1,4	0,9
20.10.76	45,1	7,2	4,6	3,6	4,0	3,5	1,0
Average	45,1	9,2	4,8	3,3	2,6	2,0	1,9

For symbols and units see Page A4.1

Table A4.5a Ammonia concentration in filtered mixed liquor of the reactors and in the effluent during first series of tests.

DATE	Inf	I	II	III	IV	V	E
5.10.76			0,0	0,0	0,0	0,0	0,0
6.10.76		5,53	0,0	0,0	0,0	0,0	0,0
7.10.76		8,0	6,0	0,0	0,0	0,0	0,0
8.10.76		8,4	0,0	0,0	0,0	0,0	0,0
9.10.76		10,6	7,2	0,0	0,0	0,0	0,0
10.10.76		11,2	6,1	0,0	0,0	0,0	0,0
11.10.76		9,6	0,0	0,0	0,0	0,0	0,0
12.10.76		9,2	0,0	0,0	0,0	0,0	0,0
13.10.76		7,7	0,0	0,0	0,0	0,0	0,0
14.10.76		10,0	2,8	0,0	0,0	0,0	0,0
15.10.76		12,0	8,0	3,0	0,0	0,0	0,0
16.10.76		9,3	4,0	0,0	0,0	0,0	0,0
17.10.76		-	-	-	-	-	0,0
18.10.76		9,3	5,0	0,0	0,0	0,0	0,0
19.10.76		8,6	4,7	0,0	0,0	0,0	0,0
20.10.76		13,0	10,0	5,0	4,0	3,4	0,0
Average		9,45	3,41	0,57	0,28	0,25	0,0

For symbols and units see Page A4.1

Table A4.6a *Nitrate + nitrite concentration in the influent, the filtered mixed liquor in the reactors and in the effluent during the first series of tests.*

DATE	Inf	I	II	III	IV	V	E
5.10.76	-	-	-	-	-	-	7,2
6.10.76	0,4	2,8	7,9	8,1	8,3	8,5	8,3
7.10.76	0,5	1,8	9,4	9,7	10,0	10,4	9,7
8.10.76	0,1	0,6	7,4	8,4	18,6	8,9	8,4
9.10.76	0,2	1,1	7,1	12,1	13,7	14,1	9,8
10.10.76	0,1	0,7	5,2	9,9	10,2	10,7	9,6
1.11.76	0,2	4,1	10,8	11,5	11,4	12,1	11,0
12.10.76	6,5	3,4	9,9	11,0	11,2	11,2	11,0
13.10.76	0,1	0,5	6,3	7,1	7,2	7,6	7,0
14.10.76	0,7	0,4	6,3	8,6	8,8	9,0	7,3
15.10.76	-	0,5	3,8	8,0	10,5	10,0	8,5
16.10.76	-	0,4	5,0	8,0	8,4	9,0	8,9
17.10.76	-	-	-	-	-	-	7,0
18.10.76	-	2,4	6,0	9,6	9,6	8,8	8,3
19.10.76	-	3,4	6,0	9,6	10,4	10,2	8,9
20.10.76	0,1	0,5	2,0	6,8	7,1	6,0	8,0
Average	0,3	1,24	5,88	9,17	9,67	9,79	8,68

For symbols and units see Page A4.1

Table A4.1b Oxygen uptake rates in the aerobic reactors, average oxygen uptake rate and temperature during the second series of tests.

DATE	I	II	III.	IV	V	X	T
21.10.76							
22.10.76		75,5	45,0	32,1	19,3	42,9	23,0
23.10.76		68,2	45,6	23,2	17,5	38,6	23,0
24.10.76		-	-	-	-	-	-
25.10.76		60,3	49,8	36,6	35,7	45,6	23,5
26.10.76		73,7	42,8	22,2	12,5	37,8	23,0
27.10.76		67,3	39,5	19,7	13,8	35,1	23,0
28.10.76		67,2	39,8	22,1	15,6	36,2	24,0
29.10.76		64,5	36,1	16,4	14,5	32,9	24,0
30.10.76		71,5	49,8	29,3	14,1	39,4	23,5
31.10.76		-	-	-	-	-	-
1.11.76		65,1	39,0	20,2	12,8	34,2	23,0
2.11.76		70,5	38,5	20,9	11,6	33,3	23,0
3.11.76		71,8	41,5	30,9	14,7	37,2	22,5
4.11.76		64,3	47,5	19,0	15,9	36,7	22,0
5.11.76		47,3	29,4	13,1	11,5	25,3	21,0
6.11.76		51,9	25,7	16,3	12,4	26,5	21,0
7.11.76		-	-	-	-	-	-
8.11.76		71,7	31,3	15,3	14,4	33,2	21,0
9.11.76		71,7	28,8	18,3	14,9	33,4	22,0
Average		66,8	38,2	21,3	14,4	35,2	22,7

X = weighted average oxygen uptake rate
T = temperature (°C)

For symbols and units see Page A4.1

Table A4.2b COD concentration of the influent, filtered mixed liquor in the reactor and effluent during the second series of tests.

DATE	Inf	I	II	III	IV	V	E
21.10.76	466						19,3
22.10.76	474	32,0	21,3	20,5	20,0	18,8	30,3
23.10.76	485						18,7
24.10.76	440	21,3	17,9	16,2	17,9	19,6	19,8
25.10.76	442						14,1
26.10.76	512	30,9	20,6	16,4	15,6	20,6	22,1
27.10.76	490						26,6
28.10.76	483	25,8	20,3	16,1	20,9	17,3	17,9
29.10.76	460						17,1
30.10.76	522	26,7	23,7	26,6	12,4	21,8	14,3
31.10.76	476						18,1
1.11.76	533	19,2	15,6	16,5	14,7	15,6	17,2
2.11.76	447						14,5
3.11.76	410	29,2	33,0	21,2	22,1	21,7	25,7
4.11.76	462						20,8
5.11.76	376	34,8	19,3	22,4	25,5	24,6	19,7
6.11.76	462						15,3
7.11.76	451	28,4	13,7	11,5	14,6	16,8	13,0
8.11.76	360						11,5
9.11.76	453	21,0	23,6	17,9	18,3	19,2	17,0
Average	461	26,9	19,9	17,9	18,9	19,6	18,5

For symbols and units see Page A4.1

Table A4.3b VSS concentrations in the reactors and average VSS concentration during the second series of tests

DATE	Inf	I	II	III	IV	V	E
21.10.76							
22.10.76		2694	2596	2696	2570	2576	2616
23.10.76							
24.10.76		2530	2452	2452	2420	2444	2460
25.10.76							
26.10.76		2816	2588	2604	2592	2564	2633
27.10.76							
28.10.76		2690	2717	2684	2706	2502	2660
29.10.76							
30.10.76		2584	2422	2362	2307	2302	2395
31.10.76							
1.11.76		2834	2684	2688	2638	2668	2703
2.11.76							
3.11.76		2588	2476	2458	2476	2452	2490
4.11.76							
5.11.76		2386	2228	2366	2538	2644	2432
6.11.76							
7.11.76		2392	2300	2287	2320	2345	2328
8.11.76							
9.11.76		2964	2802	2830	2802	2810	2840
Average		2609	2490	2501	2500	2496	2519

For symbols and units see Page A4.1

Table A4.4b TKN concentration in the influent, filtered mixed liquor of the reactors and the effluent during the second series of tests.

DATE	Inf	I	II	III	IV	V	E
21.10.76	39,3	-	-	-	-	-	1,2
22.10.76	48,0	9,7	6,8	1,8	1,2	0,9	1,2
23.10.76	39,2	8,2	4,8	2,4	2,1	1,5	1,2
24.10.76	52,8	-	-	-	-	-	1,3
25.10.76	86,6	17,1	15,6	10,0	6,2	6,7	6,8
26.10.76	80,8	11,4	7,8	3,1	2,4	1,8	1,0
27.10.76	47,3	11,3	6,9	4,8	2,0	1,8	1,9
28.10.76	46,4	11,2	6,4	3,7	3,3	2,7	1,2
29.10.76	49,2	11,2	10,7	43,2	3,4	3,4	4,5
30.10.76	55,6	13,0	9,4	4,5	3,3	3,3	2,7
31.10.76	48,2	-	-	-	-	-	6,0
1.11.76	51,4	11,0	9,7	3,8	2,4	2,5	2,3
2.11.76	55,0	8,9	5,3	4,2	2,8	3,0	1,8
3.11.76	50,1	11,2	6,8	3,6	3,2	2,6	3,3
4.11.76	42,2	10,7	7,9	5,2	3,4	2,9	3,0
5.11.76	32,9	47,7	2,4	2,7	2,6	2,2	2,4
6.11.76	37,8	58,5	3,7	3,2	3,2	2,8	2,6
7.11.76	44,3	-	-	-	-	-	1,8
8.11.76	37,5	6,0	1,8	1,8	1,8	1,4	1,4
9.11.76	43,7	7,7	4,2	2,4	3,2	3,2	2,7
Average	46,1	8,3	6,3	3,4	2,6	2,4	2,6

For symbols and units see Page A4.1

Table A4.5b *Ammonia concentration in filtered mixed liquor of the reactors and in the effluent during the second series of tests.*

DATE	Inf	I	II	III	IV	V	E
21.10.76		-	-	-	-	-	6,0
22.10.76		11,2	8,8	0,0	0,0	0,0	0,0
23.10.76		7,2	4,6	0,0	0,0	0,0	0,0
24.10.76		-	-	-	-	-	0,0
25.10.76		16,0	15,3	10,4	5,6	4,9	5,1
26.10.76		9,0	5,1	0,0	0,0	0,0	0,0
27.10.76		4,9	3,2	0,0	0,0	0,0	0,0
28.10.76		10,0	6,0	0,0	0,0	0,0	0,0
29.10.76		11,0	5,1	0,0	0,0	0,0	0,0
30.10.76		10,9	8,8	0,0	0,0	0,0	0,0
31.10.76		-	-	-	-	-	0,0
1.11.76		9,6	7,0	0,0	0,0	0,0	0,0
2.11.76		8,8	5,3	0,0	0,0	0,0	0,0
3.11.76		9,0	5,3	0,0	0,0	0,0	0,0
4.11.76		9,1	6,0	0,0	0,0	0,0	0,0
5.11.76		3,0	0,0	0,0	0,0	0,0	0,0
6.11.76		9,0	0,0	0,0	0,0	0,0	0,0
7.11.76		-	-	-	-	-	0,0
8.11.76		3,3	0,0	0,0	0,0	0,0	0,0
9.11.76		4,0	0,0	0,0	0,0	0,0	0,0
Average		7,7	4,3	0,0	0,0	0,0	0,0

For symbols and units see Page A4.1

Table A4.6b Nitrate + nitrite concentration in the influent, the filtered mixed liquor of the reactors and the effluent during the second series of tests.

DATE	Inf	I	II	III	IV	V	E
21.10.76	0,3	-	-	-	-	-	7,5
22.10.76	0,2	0,5	2,5	9,0	9,3	9,6	7,3
23.10.76	0,1	0,4	2,7	6,0	6,8	7,0	5,5
24.10.76	0,2	-	-	-	-	-	2,4
25.10.76	0,1	0,5	1,3	7,0	11,7	12,8	9,4
26.10.76	0,1	1,1	3,1	7,1	7,3	8,0	7,3
27.10.76	0,2	3,0	4,5	6,8	6,9	6,9	6,0
28.10.76	0,1	0,6	2,0	6,5	6,6	7,0	6,3
29.10.76	0,1	1,3	4,3	7,5	7,3	7,8	6,5
30.10.76	0,1	1,2	2,9	8,7	8,6	8,7	7,0
31.10.76	0,1	-	-	-	-	-	6,0
1.11.76	0,1	1,5	3,8	8,7	8,6	8,6	8,0
2.11.76	0,1	2,5	5,4	8,7	8,9	9,2	8,5
3.11.76	0,2	1,0	3,8	7,2	7,4	7,9	7,1
4.11.76	0,1	2,3	3,2	8,0	8,1	8.6	7,9
5.11.76	0,1	1,8	7,1	7,3	7,6	7,8	7,8
6.11.76	0,1	8,1	7,7	8,7	9,0	8,4	8,4
7.11.76	0,1	-	-	-	-	-	8,8
8.11.76	0,1	1,5	8,5	9,1	9,2	9,2	8,0
9.11.76	0,1	2,7	6,2	7,4	7,8	7,7	7,0
Average	0,1	1,7	4,3	7,7	8,2	8,45	7,43

For symbols and units see Page A4.1

APPENDIX A5

RESULTS AND SIMULATIONS OF THE EXPERIMENTS USING THE SINGLE REACTOR SYSTEM UNDER ALTERNATING ANOXIC AND AEROBIC CONDITIONS

APPENDIX A5

This appendix contains the experimental data of the tests carried out on the single reactor system under alternating anoxic and aerobic periods; described in Chapter 4, Section 7.1. A total of 19 tests were carried out. The investigation was divided into two parts : In the first part - composed of 13 tests - there was a single anoxic period per day. The following anoxic periods were investigated :

2 h anoxic per day : test 1 to 3
3 h anoxic per day : test 4 to 6
4 h anoxic per day : test 7 to 9
5 h anoxic per day : test 10 to 13

Total 13 tests

In the second part the system was operated with more than one anoxic period per day. This part is composed of six tests. The following tests were done :

cyclic 1 h anoxic followed by 2 h aerobic : test 14 to 16
cyclic 2 h anoxic followed by 2 h aerobic : test 17 to 19

Total 6 tests

The experimental values of the measurable variables in the reactor for all the tests are given in Tables A5.1 to A5.19.

The simulated and experimental values of the measurable variables are shown plotted in Fig A5.1 to A5.19.

The following conventions apply to Tables A5.1 to A5.19 -

APPENDIX A5(continued)

Column 1 : Sample : 100 ml samples were taken from the influent (Infl.) and effluent(Effl.) flows. A filtered effluent sample (Effl. filter) was also analysed. In addition, ten mixed liquor samples were taken at various times during an experiment and filtered immediately.

Column 2 : Time : Time of the day a sample was taken and/or a test of the oxygen uptake rate was done.

Column 3 : OUR : Oxygen uptake rate in mg $O_2.\ell.^{-1}h^{-1}$ measured at the times indicated in column 2.

Column 4 : COD : COD concentrations in mg COD.ℓ^{-1} of samples in column 1 at times in column 2.

Column 5 : TKN : TKN concentrations in mgN.ℓ^{-1} of samples in column 1 at times in column 2.

Column 6 : NH_3 : Ammonia concentrations in mg N.ℓ^{-1} of samples in column 1 at times in column 2.

Column 7 : Alk : Alkalinity in mg $CaCO_3.\ell^{-1}$ of samples in column 1 at times in column 2.

Column 8 : NO_3 : Nitrate concentrations in mg N.ℓ^{-1} of samples in column 1 at times in column 2.(nitrite concentrations were normally less than 0,5 mg N.ℓ^{-1}).

Column 9 : VSS : Mixed liquor volatile suspended solids concentration in mg VSS.ℓ^{-1} of samples in column 1 taken at times in column 2.

Column 10: pH : Mixed liquor pH at times in column 2.

Temperature for all tests : (20.0 $\pm$ 0,2)°C.

Table A5.1: Test 1: 2 hours anoxic period per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			500	43,4	18,7	319	-		7,8
Effl.			65	4,5	0,5	133	25,8		7,5
Effl. (Filter)			77	4,6	0,5	133	25,8		-
1	9.00	49,0	69	6,3	1,2	141	24,0		7,2
2	10.00	-	69	3,4	< 0,5	131	25,0		-
3	12.00*	46,5	69	4,2	< 0,5	139	24,8	2591	-
4	13.00	-	69	8,2	2,5	167	19,5		-
5	14.00**	64,5	75	10,5	5,3	193	15,5	2502	7,3
6	14.30	70,5	75	9,1	3,2	170	18,2		-
7	15.00	65,5	-	7,3	1,4	158	21,0		-
8	16.00	50,5	73	3,6	< 0,5	148	21,8	2424	7,2
9	17.00	49,5	-	4,2	< 0,5	147	22,5		-
10	18.00	-	-	3,6	< 0,5	146	23,0		-
	14.15	65,0							
	15.30	52,5							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 10.6.80

Table A5.2: Test 2: 2 hours anoxic period per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			512	43,4	19,8	315	-		7,8
Effl.			69	4,2	1,1	135	24,8		7,6
Effl. (Filter)			69	4,4	1,1	135	24,8		-
1	7.60	48,5	73	3,5	< 0,5	130	25,8		7,2
2	8.00	-	73	3,5	< 0,5	130	25,8		7,2
3	9.00*	46,5	69	-	< 0,5	149	23,5	2415	-
4	10.00	-	73	6,7	3,0	173	20,2		-
5	11,00**	-	69	9,4	5,5	204	13,8	2375	7,3
6	11.30	62,5	77	8,4	3,3	188	17,5		7,3
7	12.00	62,5	73	7,0	1,8	173	21,8		7,2
8	13.00	49.0	69	3,9	1,0	147	22,8	2286	-
9	14.00	47,5	69	4,3	< 0,5	146	24,0		-
10	15.00	46,5	61	3,5	< 0,5	145	24,0		-
	11.10	69,5							
	12.10	62,0							
	12.20	60,5							
	12.30	61,5							
	12.40	59,0							
	13.30	50,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 11.6.80

Table A5.3: Test 3: 2 hours anoxic period per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			512	44.0		317			7,8
Effl.			73	4,4		135	26,0		7,6
Effl. (Filter)			69	4,4		135	25,0		
1	7,00		73	3,8	< 0,5	130	24,2		7,3
2	8,00	49,0	69	3,4	< 0,5	125	24,8		7,3
3	9,00*	49,0	81	3,5	< 0,5	125	25,0	2498	
4	10,00		73	7,0	3,2	169	19,5		
5	11,00**	51,5	73	10,1	5,5	195	13,5	2456	7,4
6	11,30	64,5	69	8,4	3,6	180	18,0		
7	12,00	64,5	77	5,6	1,8	165	21,5		7,4
8	13,00	49,0	69	3,6	0,6	149	24,6	2517	
9	14,00	50,5	73	3,5	< 0,5	151	22,8		
10	15,00	49,0	73	3,7	< 0,5	145	24,0		7,3
	11,15	64,5							
	12,30	57,5							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 12.6.80

Table A5.4: Test 4: 3 hours anoxic period per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			516	48,3	40,0	301			7,8
Effl.			86	6,7	1,2	100	29,8		7,3
Effl. (Filter)			71	7,0	1,2	100	29,4		
1	9,20	47,5	84	4,9	0,6	78	33,2	2459	
2	10,00*	48,5		7,0	0,5	74	32,5	2458	6,95
3	11,00			11,5	5,6	111	25,5		7,15
4	12,00			15,7	8,5	147	18,6		7,35
5	13,00**	55,0	92	15,1	13,6	180	14,7		7,4
6	13,30	63,5	64	16,1	13,4	164	17,0		7,3
7	14,00	63,5		13,7	9,2	161	16,8		7,3
8	15,00	84,0		11,9	5,3	121	24,9		7,1
9	16,00	59,0		5,6	1,2	104	26,7		7,1
10	17,00	47,5		2,8	0,6	93	29,4		7,15
	13,15	60,0							
	13,45	64,0							
	14,20	63,0							
	14,40	62,0							
	15,30	60,5							
	16,30	51,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 14.5.80

Table A5.5: Test 5: 3 hours anoxic period per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			482	48,3	41,6	303			7,8
Effl.			65	5,9		103	29,5		7,4
Effl. (Filter)			61	5,6		101	29,4		
1	9,00	47,0	84	5,9	1,5	92	29,0		7,1
2	10,00*	48,0		4,5	1,1	85	30,8	2465	
3	11,00			7,0	5,2	123	22,7		
4	12,00			12,9	8,5	150	17,8		
5	13,00**	55,0	84	15,4	9,8	178	13,5		7,2
6	13,30	63,0	75	13,3	8,6	166	21,4		
7	14,00	59,0		11,9	5,7	156	20,5		
8	15,00	59,0		6,3	3,7	111	29,5		
9	16,00	47,0		4,2	0,8	102	26,0		
10	17,00	46,0		3,6	< 0,5	99	26,8		
	13,15	63,0							
	13,45	63,0							
	14,15	63,0							
	14,30	61,5							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 15.5.80

Table A5.6: Test 6: 3 hours anoxic period per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			541	48,2	32,4	313			7,8
Effl.			78	5,6	1,1	107	29,5		7,4
Effl. (Filter)			65	4,0	1,1	101	29,4		
1	9,00	50,5	76	4,9	1,3	102	27,8	2552	6,95
2	10,00*	48,0	71	4,5	1,2	82	32,8	2598	7,1
3	11,00		96	9,4	4,7	120	26,8		
4	12,00		92	13,3	6,6	147	20,2		
5	13,00**	59,0	98	17,5	9,7	177	14,5		
6	13,30	63,5	86	12,0	8,6	164	17,8		
7	14,00	65,0		8,1	5,8	146	19,4		
8	15,00	60,5	82	6,4	2,4	120	23,4		
9	16,00	51,0		4,5	0,7	100	28,0		
10	17,00	50,5	77			96	28,2		
	14,40	63,0							
	15,30	62,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 16.5.80

Table A5.7: Test 8: 4 hours anoxic per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			540	54,8	25,0	323			7,8
Effl.			77	6,3	1,5	118	28,0		7,4
Effl. (Filter)			69	6,3	1,5	117	28,0		
1	9,00	49,0	71	5,6	0,9	102	30,0		7,0
2	10,00*		69	6,3	0,7	92	30,2	2558	7,0
3	11,00		97	9,1	4,9	133	24,2	2540	7,1
4	12,00		77	11,9	7,9	168	16,8	2537	7,2
5	13,00		81	15,8	11,3	201	11,2		7,3
6	14,00**	58,5	89	20,0	13,3	226	6,6	2587	7,4
7	14,30	67,5	81	17,5	11,7	201	11,2		7,4
8	15,00	69,0	77	13,3	8,6	181	14,8		7,3
9	16,00	65,0	77	9,1	4,0	142	19,0		7,2
10	17,00	53,0	69	5,6	0,7	107	25,0		
	14,15	67,0							
	15,30	69,0							
	16,25	65,0							
	16,40	64,0							
	16,50	59,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 28.5.80

Table A5.8: Test 7: 4 hours anoxic per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			520	54,9	21,0	323			7,8
Effl.			73	5,6	1,4	114	27,8		7,5
Effl. (Filter)			57		1,4	113	27,8		
1	9,00		70	6,7	0,8	99	30,2	2396	
2	10,00 *	47,0	62	5,6	0,8	100	29,8		7,2
3	11,00		82	12,3	4,0	134	24,6	2522	
4	12,00		86	17,2	7,7	176	16,4		
5	13,00		86	17,5	10,9	202	12,8	2483	
6	14,00 **	59,0	78		12,6	226	7,0		7,4
7	15,00	64,5	62	14,0	7,1	173	15,4	2456	
8	16,00	65,0	62	8,4	3,2	138	21,5		
9	17,00	49,0	62	5,6	0,8	113	26,0	2397	
10	18,00		62	4,9	< 0,5	107	27,5		
	14,15	62,0							
	14,30	61,5							
	14,45	61,5							
	15,15	63,0							
	15,30	61,0							
	15,45	62,5							
	16,15	58,0							
	16,30	62,5							
	16,45	50,0							
	17,15	49,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 29.5.80

Table A5.9: Test 9: 4 hours anoxic per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			550	55,0	33,0	325			7,8
Effl.			85	6,7	2,1	121	29,0		7,5
Effl. (Filter)			71	6,9	2,1	121	28,5		
1	9,00			6,3	0,7	95	29,8		7,0
2	10,00*	53,0	n	5,9	0,7	95	30,8	2447	7,0
3	11,00		o	10,9	4,9	132	23,8		7,1
4	12,00		t	13,3	8,4	172	17,2		7,2
5	13,00			16,8	11,9	199	9,8	2564	7,3
6	14,00**	59,0	d	18,2	14,7	238	6,0		7,4
7	14,30	64,0	o	15,7	12,4	216	9,6		7,4
8	15,00	63,0	n	12,9	7,7	190	14,0		7,3
9	16,00	65,0	e	9,5	5,3	150	18,7	2491	7,3
10	17,00	59,5		5,9	1,1	121	24,6		
	14,15	66,0							
	15,30	60,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 30.5.80

Table A5.10: Test 10: 5 hours anoxic per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			475	43,4	22,0	315			
Effl.			77	5,3	3,0	156	21,7		
Effl. (Filter)			69	5,3	2,5	154	21,9		
1	9,00*	47,0	73	5,6	0,6	125	23,8		
2	10,00		69	7,7	4,6	165	18,5	2533	7,3
3	11,30		93	12,6	8,1	209	12,0		
4	13,00		77	15,4	13,4	247	2,5	2469	
5	14,00**	59,0	138	17,1	13,9	261	1,0	2397	7,6
6	14,30	65,0	77	14,0	10,5	235	4,4		7,5
7	15,00	64,0	81	11,5	8,6	215	8,5	2335	
8	16,00	66,0	81	7,7	3,4	173	15,6		
9	17,15	55,0	79	5,6	0,8	140	19,8	2354	
10	18,00	47,0	85	4,5	< 0,5	140	20,3		
	14,15	68,0							
	15,30	62,0							
	16,30	63,0							
	17,30	44,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 5.6.80

Table A5.11: Test 11: 5 hours anoxic per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			461	44,8	22,0	304			7,8
Effl.			66	5,0	2,2	148	20,2		7,4
Effl. (Filter)			69	5,7	2,1	148	19,8		
1	8,00	46,0	66	4,9	1,1	120	26,0		7,3
2	9,00*	47,0	66	4,9	1,1	125	24,5	2519	
3	11,00		74	9,5	5,1	177	14,9		
4	13,07		74	15,2	10,0	229	2,8		7,4
5	14,00**	54,0	78	17,9	13,0	247	1,0	2477	7,6
6	14,30	66,0	74	14,0	9,3	215	5,2		7,4
7	15,00	62,0	88	11,2	7,6	201	8,6	2367	7,4
8	16,00	66,0	74	7,3	3,3	163	14,9		
9	17,00	48,0	69	3,9	0,5	136	18,9	2329	7,3
10	18,00	40,0	74				19,2		
	14,15	66,0							
	15,30	66,0							
	16,30	51,5							
	17,30	45,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 6.6.80

Table A5.12: Test 12: 5 hours anoxic per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			493	35,1	15,2	302			7,8
Effl.			59	5,3	1,3	184	16,2		7,6
Effl. (Filter)			73	5,2	1,3	184	16,0		
1	8,00	39,5	62	4,5	1,3	175	16,0		
2	9,00*	41,5	57	4,5	0,6	168	18,8	2559	
3	11,00		82	10,1	3,9	210	9,0		
4	12,00		66	9,1	5,7	240	4,6	2528	
5	13,00		78	11,5	6,5	258	1,0		
6	14,00**	48,0	78	12,9	8,8	254	1,0	2445	
7	14,30	68,0	62	8,9	8,6	235	3,2		
8	15,00	66,0	62	5,6	3,5	216	7,8		
9	16,00	48,0	57	3,1	0,6	190	14,0		
10	17,00	45,0	74	5,3	< 0,5	178	13,5	2376	
	14,10	66,0							
	14,20	66,0							
	15,30	63,0							
	16,30	44,0							
	18,00	44,0							

* **begin anoxic period**

** **end anoxic period**

For symbols and units see page A5.1

Date: 7.6.80

Table A5.13: Test 13: 5 hours anoxic per day

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			480	35,1	15,2	303			7,8
Effl.			73	5,3		196	15,2		7,5
Effl. (Filter)			73	5,2		192	15,8		
1	8,00	39,0	67	4,5	0,6	171	17,5		7,4
2	9,00*	41,0	67	3,4	0,5	172	18,5		
3	10,30		79	9,1	3,7	210	11,5	2539	
4	12,00			10,1	5,9	241	4,6		
5	13,00		83	11,5	8,9	257	1,0	2513	
6	14,00**		79	14,7	9,4	261	1,0		7,8
7	14,30	60,0		11,5	8,9	263	1,0		7,7
8	15,00	66,0	71	6,7	3,3	221	7,8		
9	16,00	55,0	67	4,2	1,6	192	14,3		
10	17,00	47,0		3,6	< 0,5	185	16,2	2487	
	14,10	63,0							
	14,20	66,0							
	15,30	63,0							
	16,30	48,0							
	16,45	47,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 8.6.80

Table A5.14: Test 15: 1 hour anoxic - 2 hours aerobic cyclic

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			526	43,4	21,8	311			7,8
Effl.			53	3,5	1,5	155	15,7		7,4
Effl. (Filter)			61	3,5	1,5	155	15,7		
1	9,00*		63	4,2	1,3	161	15,0		7,2
2	10,00**	62,0	59	5,6	4,5	183	11,6	2715	7,2
3	10,30	57,0		5,6	1,9	167	15,0		7,2
4	11,00	64,0		2,2	1,3	156	17,0	2811	7,2
5	12,00*	49,0	67	2,8	0,7	149	19,2		7,2
6	13,00**	73,0	63	6,3	3,4	191	10,5	2613	
7	13,30	66,0		4,9	2,0	173	14,4		
8	14,00	58,0		4,2	1,3	161	17,0	2537	7,2
9	15,00*	48,5	63	4,9	0,7	153	19,3		
10	16,00**	65,0	71	6,7	3,4	188	12,8	2482	7,2
	10,15	62,0							
	10,45	62,0							
	11,15	53,0							
	11,30	45,0							
	14,20	56,0							
	14,40	51,0							
	16,30	65,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 23.6.80

Table A5.15: Test 14: 1 hour anoxic - 2 hours aerobic cyclic

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			496	42,7		141			7,4
Effl.			69	4,2		37	14,0		7,0
Effl. (Filter)			69	4,5			14,0		
1	7,00	57,0		3,5	1,0	28	16,0		6,4
2	8,00*	58,0	n	4,7	0,8	22	17,5	2813	6,4
3	9,00**	66,0	o	7,0	3,6	53	10,6		6,8
4	9,30	66,0	t	5,5	2,1	39	12,8	2751	6,6
5	10,00	51,0		3,5	1,4	32	14,8		6,5
6	11,00*	45,0	d	3,8	1,2	22	16,0	2720	6,4
7	12,00**	47,0	o	6,6	4,0	53	9,6		6,9
8	12,30	62,0	n	3,5		40	11,6	2452	6,8
9	13,00	57,0	e	3,5	2,2	32	13,2		6,5
10	14,00	51,0		3,5		19	16,0	2552	6,4
	7,30	59,0							
	10,20	56,0							
	10,45	55,0							
	12,10	52,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 24.6.80

Table A5.16: Test 16: 1 hour anoxic - 2 hours aerobic cyclic

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			524	43,4	21,8	314			7,8
Effl.			53	3,6	1,5	168	16,2		7,5
Effl. (Filter)			53	3,6	1,5	168	16,2		
1	5,30*	47,0	51	4,2	0,8	149	18,2		7,2
2	6,30**	75,0	54	6,7	4,0	181	13,9	2681	
3	7,00	44,0		5,5	1,9	166	16,0		7,2
4	7,30	45,0		4,2	0,7	152	18,2	2630	
5	8,30*	49,0	51	4,2	0,6	147	20,8		
6	9,30**	66,0	71	6,3	4,6	188	12,1	2512	
7	10,00	61,0		5,0	2,3	165	14,8		7,2
8	10,30	59,0		3,5	1,2	157	18,0	2533	
9	11,30*	50,0	51	2,8	0,6	150	20,5		
10	12,30**	60,0	51	6,7	3,3	187	15,2	2494	
	6,45	55,0							
	7,15	49,0							
	9,45	51,0							
	10,20	52,0							
	11,00	48,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 25.6.80

Table A5.17: Test 17: 2 hours anoxic - 2 hours aerobic cyclic

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			496	46,5	32,4	311			7,8
Effl.			57	7,3		187	10,0		7,6
Effl. (Filter)			57	7,0	4,4	189	10,2		
1	9.30*		65	7,0	1,5	161	10,3		7,2
2	10,30			8,4	5,2	197	5,0	2967	7,3
3	11,30**	52,0	63	11,5	7,9	223	0,0		7,5
4	12,30	69,0		6,7	3,4	184	9,2	2839	7,4
5	13,30*	63,0	77	3,9	1,0	152	15,0		7,3
6	14,30			8,1	4,6	192	8,0	2860	7,3
7	15,30**	70,0	65	11,5	5,9	221	0,0		7,5
8	16,30	62,6		7,0	3,5	187	8,0		7,4
9	17,30*		61	4,2	1,5	159	13,1		7,3
10	19,30**		65	11,2	6,4	226	0,0	2781	

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date 26.6.80

Table A5.18: Test 18: 2 hours anoxic - 2 hours aerobic cyclic

Sample	Time	OUR	COD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			512	46,6	30,0	315			7,8
Effl.			58	7,7	4,3	192	7,2		7,5
Effl. (Filter)			57	7,7		193	7,1		
1	9,30*	67,0	67	6,3	2,3	166	11,5		
2	10,30			9,1	7,3	197	6,1	2987	
3	11,30**	66,0	75	11,9	8,6	227	<0,5		
4	12,30	67,0		7,7	5,1	187	7,6	2987	
5	13,30*	66,0	59	4,9	2,3	171	9,7		
6	14,30			8,4	5,8	194	6,6	2925	
7	15,30**	55,0	67	10,9	8,6	232	<0,5		
8	16,30	69,0		7,0	5,7	187	7,6	2967	
9	17,30*	68,0	63	4,9	2,3	155	13,0		
10	19,00**		63	10,9	5,7	212	3,0	2953	
	11,45	57,0							
	12,00	75,0							
	13,00	71,0							
	15,45	70,0							
	16,00	69,0							
	17,00	69,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 27.6.80

Table A5.19: Test 19: 2 hours anoxic - 2 hours aerobic cyclic

Sample	Time	OUR	GOD	TKN	NH_3	ALK	NO_3	VSS	pH
Infl.			492	49,0	33,5	245			7,8
Effl.			62	7,1	3,5	114	7,1		7,5
Effl. (Filter)			54	6,9	3,5	114	5,8		
1	9,30*	68,0	66	4,6	1,1	80	12,5		7,1
2	10,30			9,2	4,0	115	6,4	2822	7,2
3	11,30**	69,0	70	11,5	6,6	147	0,0		7,4
4	12,30	69,0		7,7	2,8	102	8,2	2773	7,2
5	13,30*	70,0	59	5,6	0,7	70	14,6		7,1
6	14,30			8,8	4,0	118	5,5	2832	7,2
7	15,30**	72,0	67	8,6	6,6	150	0,5		7,4
8	16,30	69,0		6,3	2,5	109	7,5		
9	17,30*	70,0	67	6,3	0,7	75	15,5		
10	19,30**	70,0	69	11,3	6,6	153	0,5	2692	
	11,45	67,0							
	12,15	72,0							
	13,00	70,0							
	17,00	71,0							

* begin anoxic period

** end anoxic period

For symbols and units see page A5.1

Date: 28.6.80

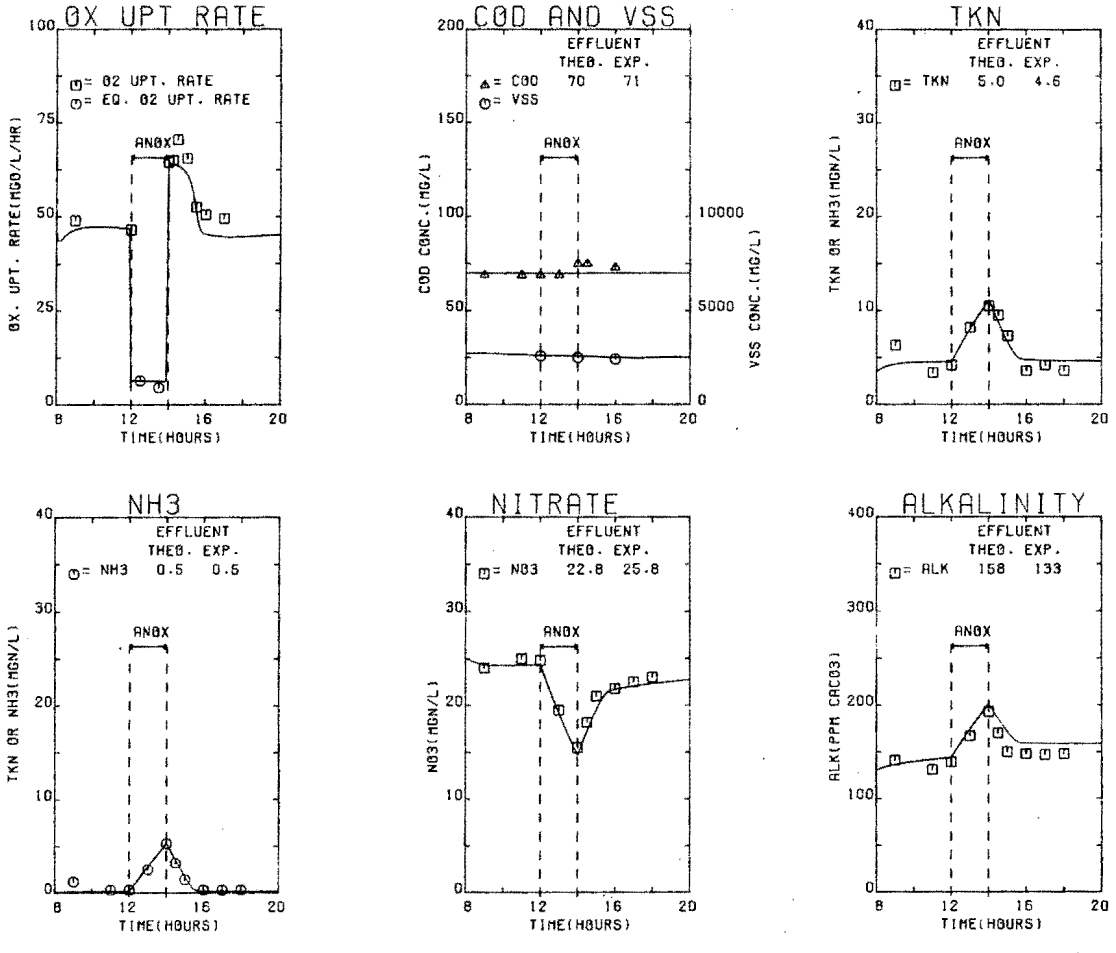

Fig. A5.1 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 2 hours per day. (Experimental data in Table A5.1)*

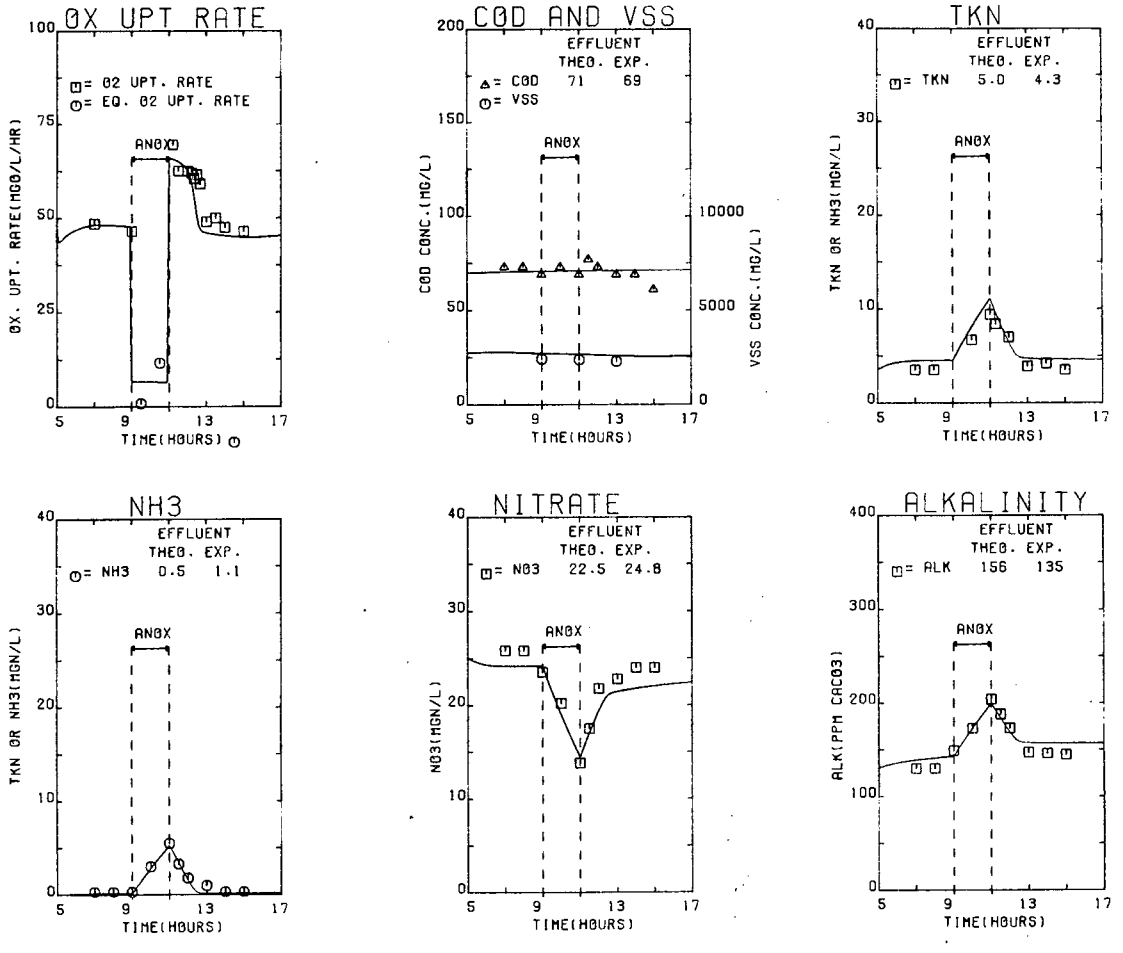

Fig.A5.2 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 2 hours per day. (Experimental data in Table A5.2)*

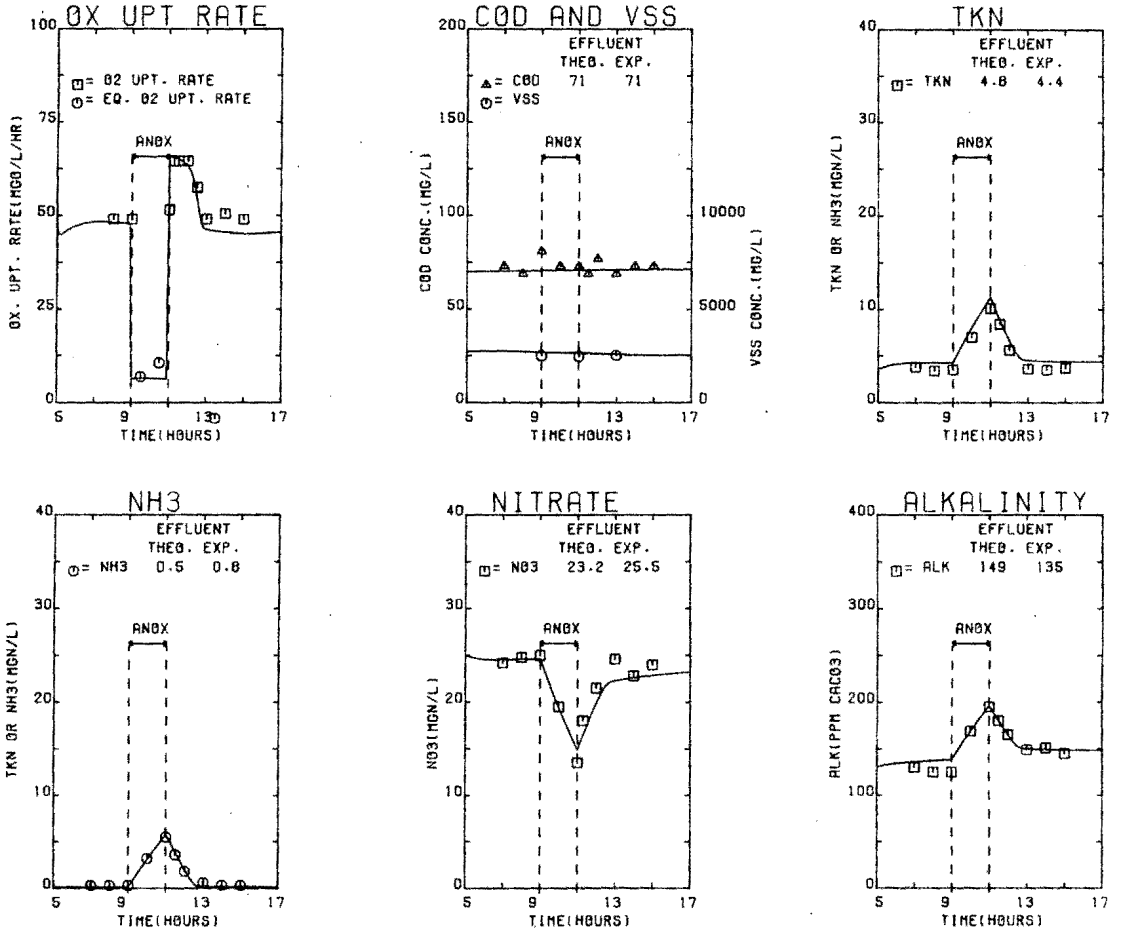

Fig.A5.3 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 2 hours per day. (Experimental data in Table A5.3)*

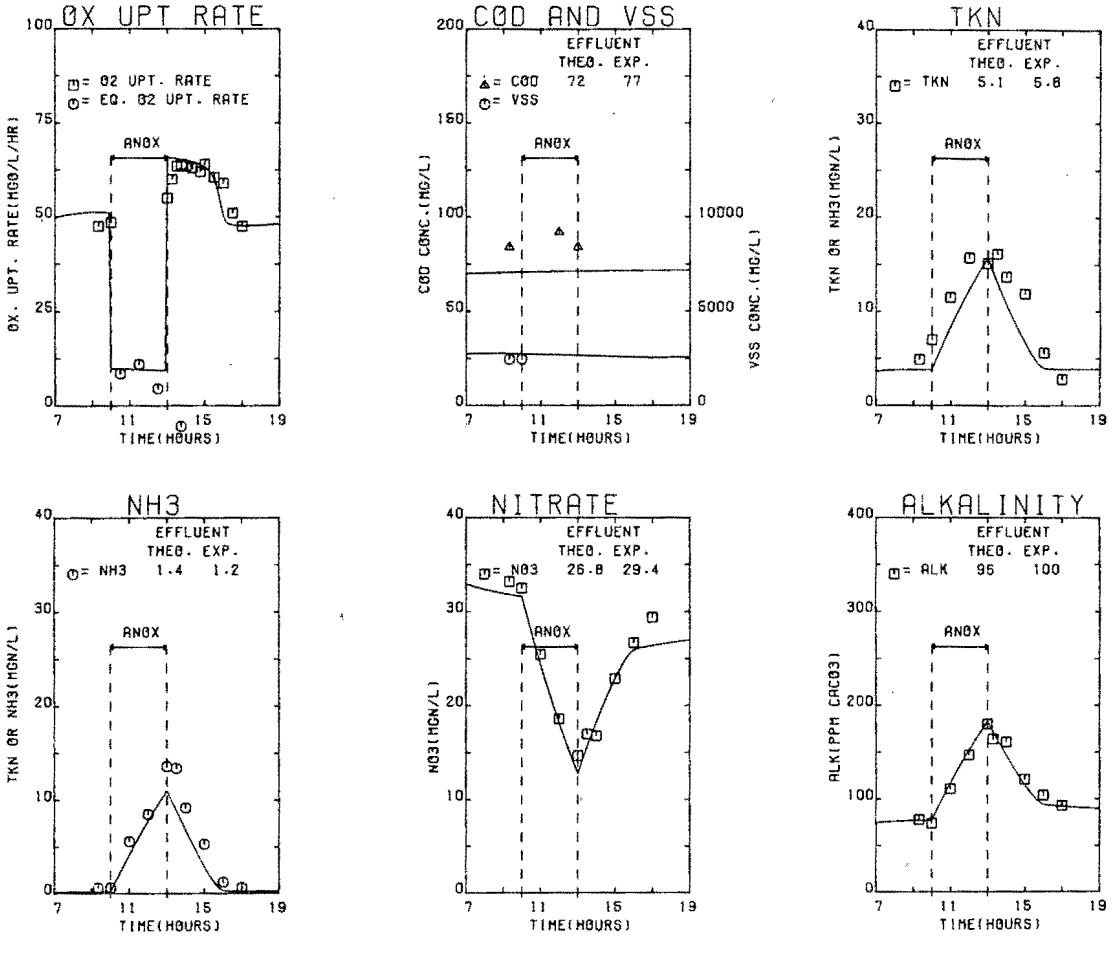

Fig. A5.4 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 3 hours per day. (Experimental data in Table A5.4)*

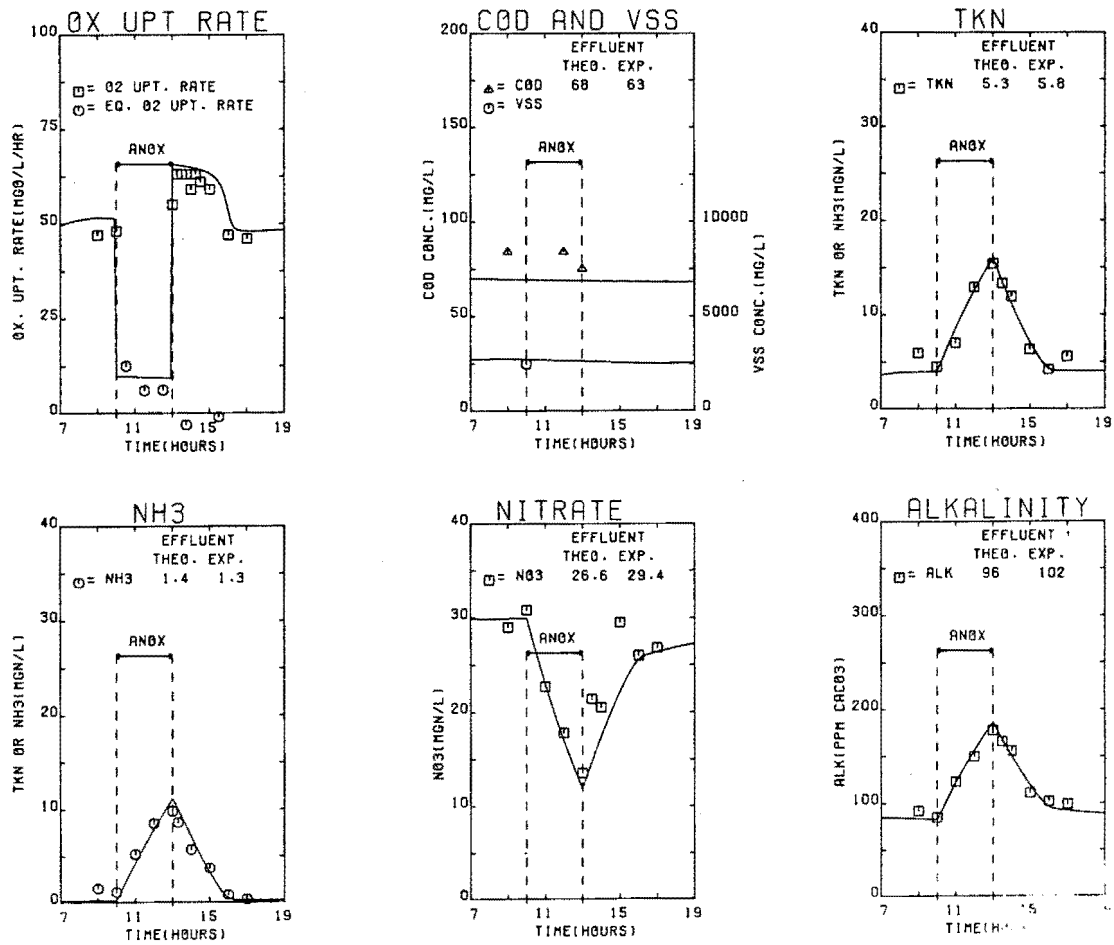

Fig. A5.5 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 3 hours per day. (Experimental data in Table A5.5)*

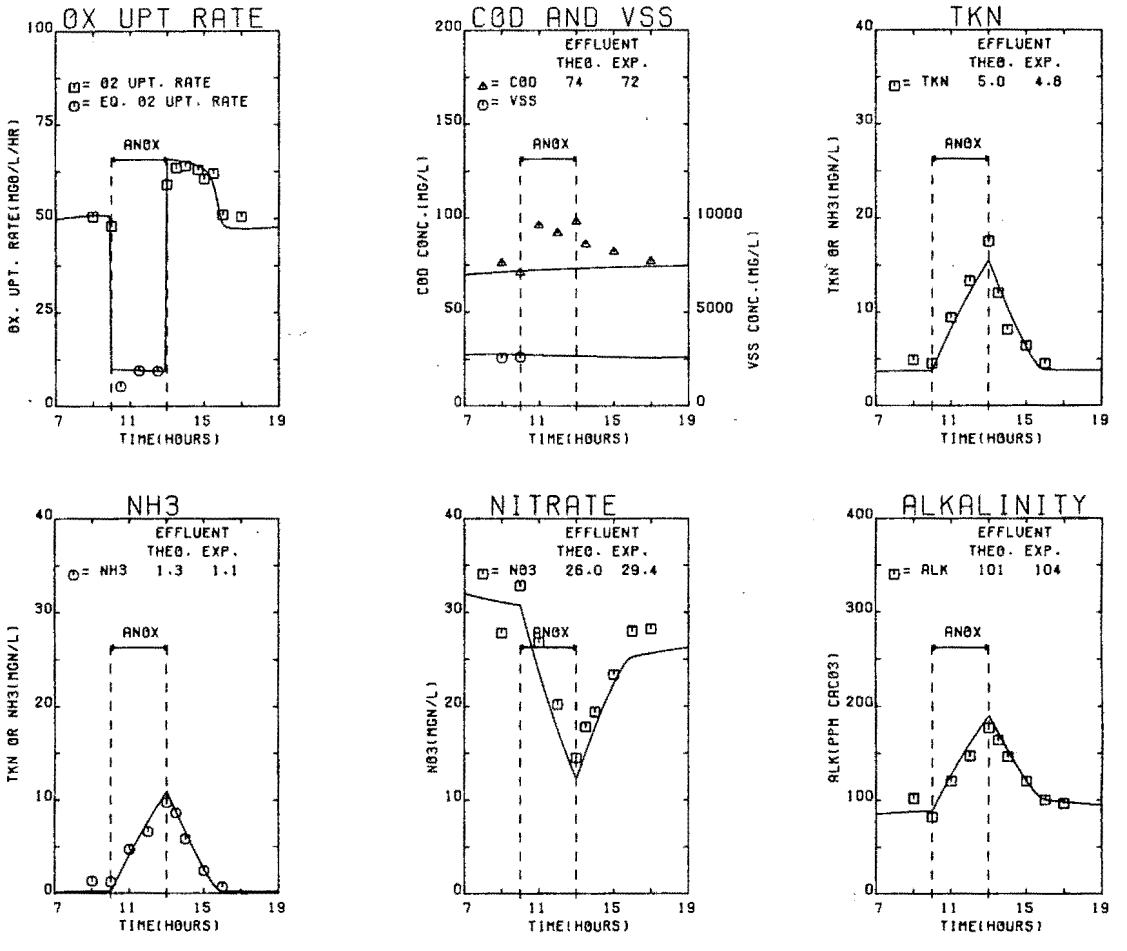

Fig. A5.6 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 3 hours per day. (Experimental data in Table A5.6)*

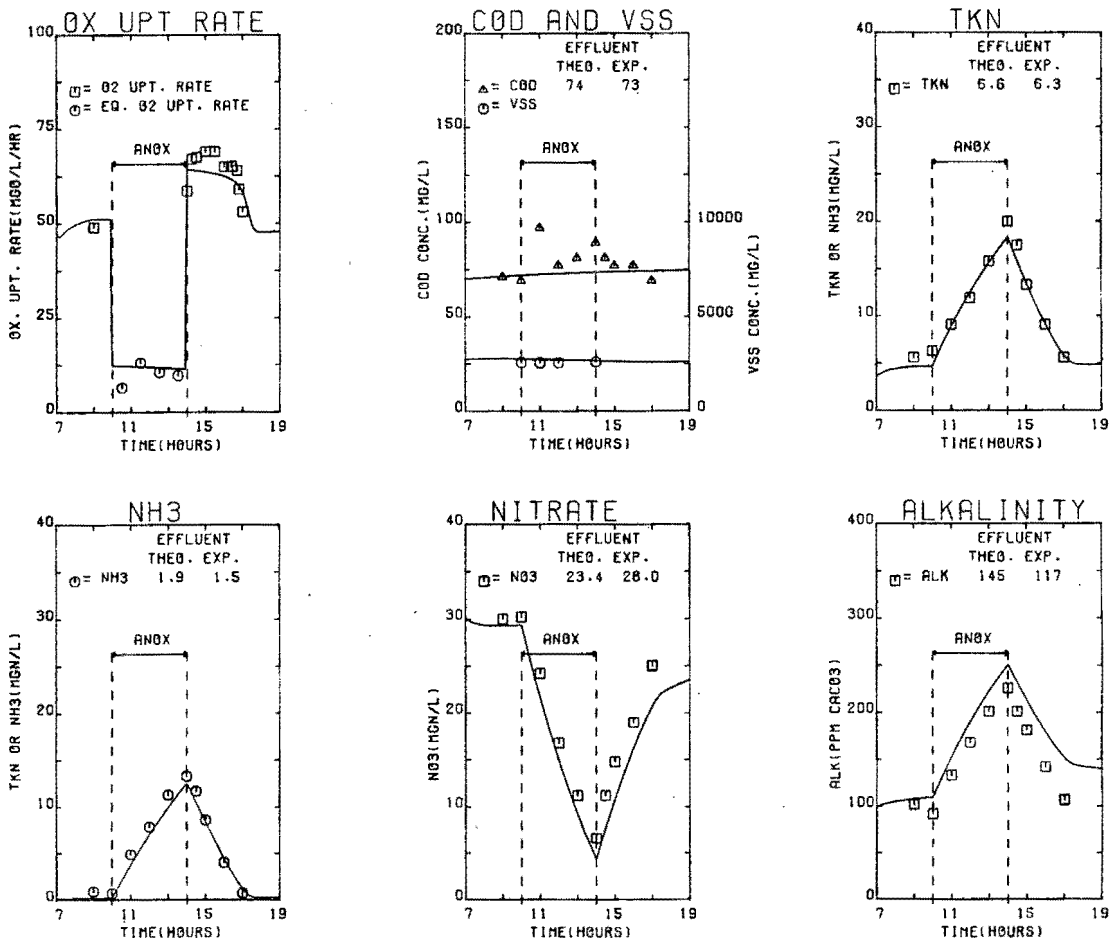

Fig.A5.7 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 4 hours per day. (Experimental data in Table A5.8)*

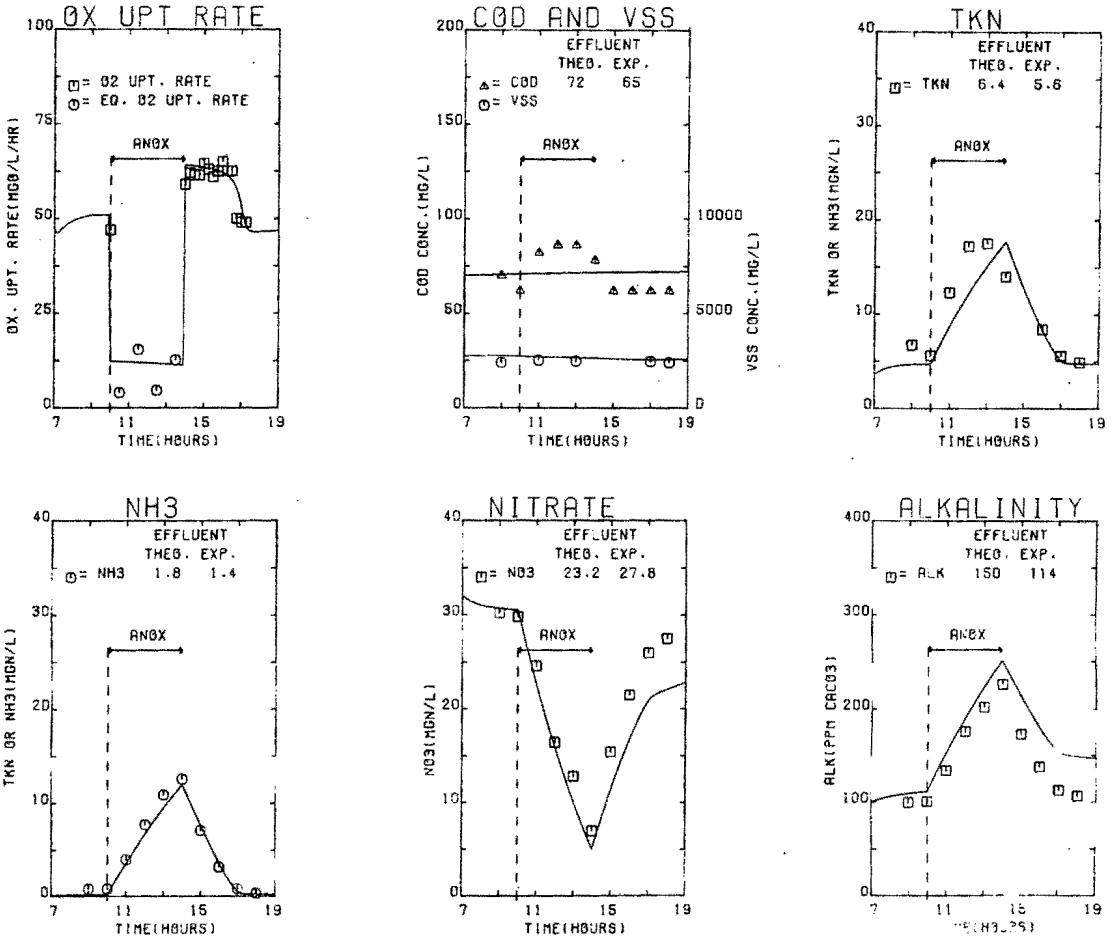

Fig. A5.8 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 4 hours per day. (Experimental data in Table A5.7)*

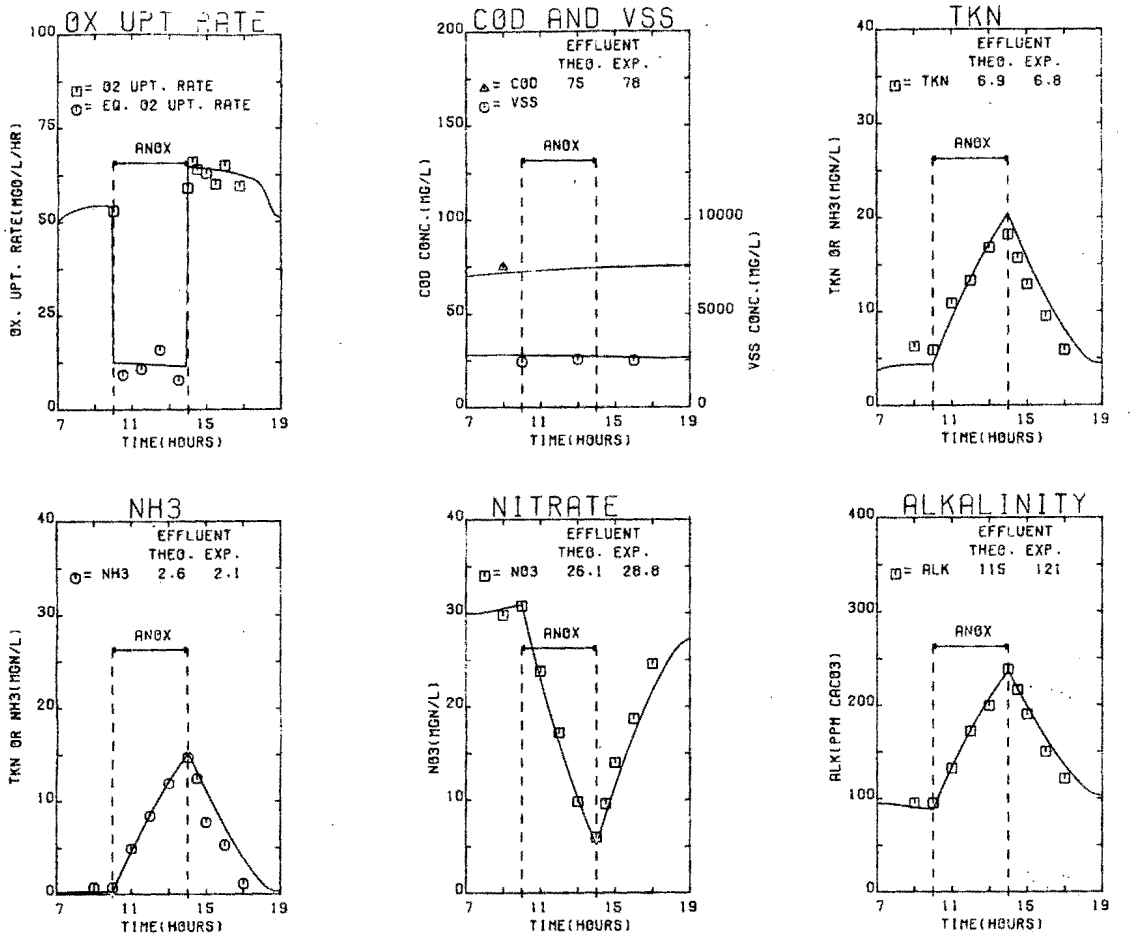

Fig. A5.9 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 4 hours per day. (Experimental data in Table A5.9)*

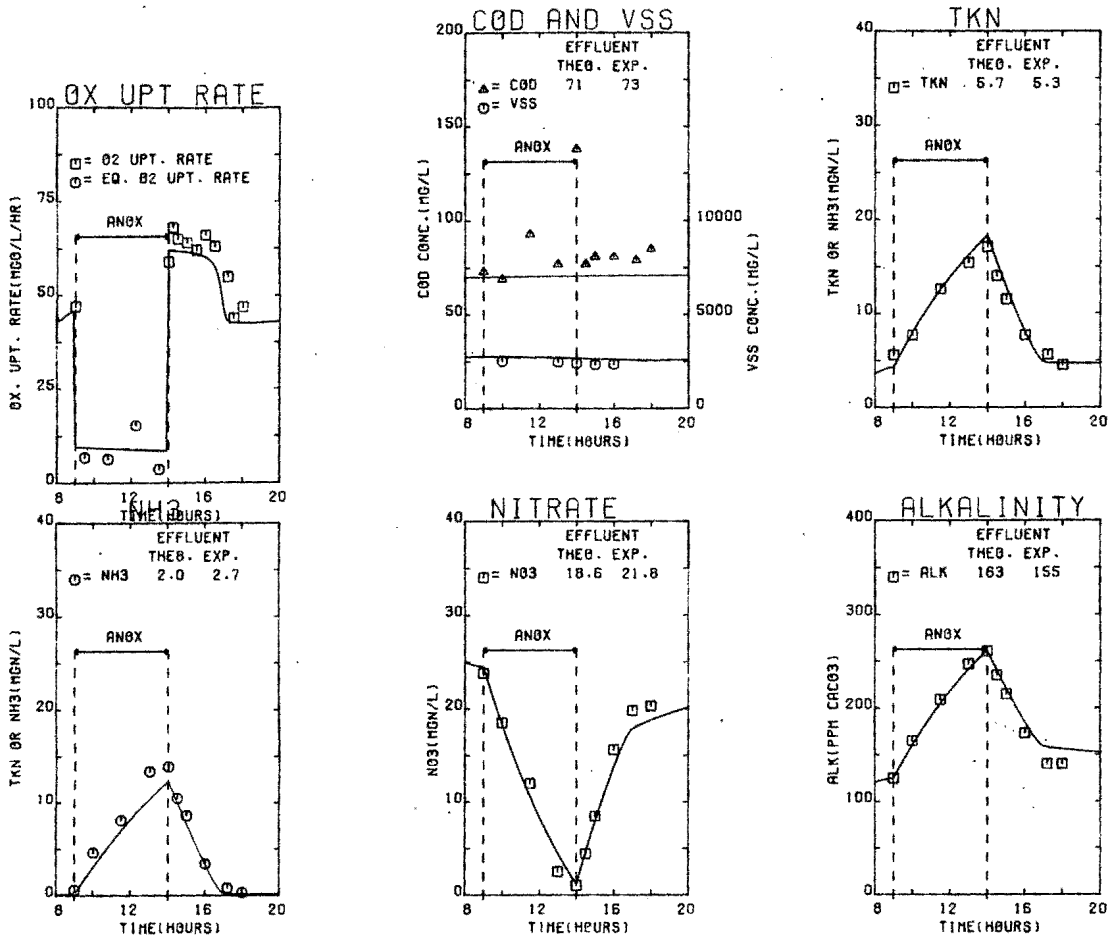

Fig.A5.10 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 5 hours per day. (Experimental data in Table A5.10)*

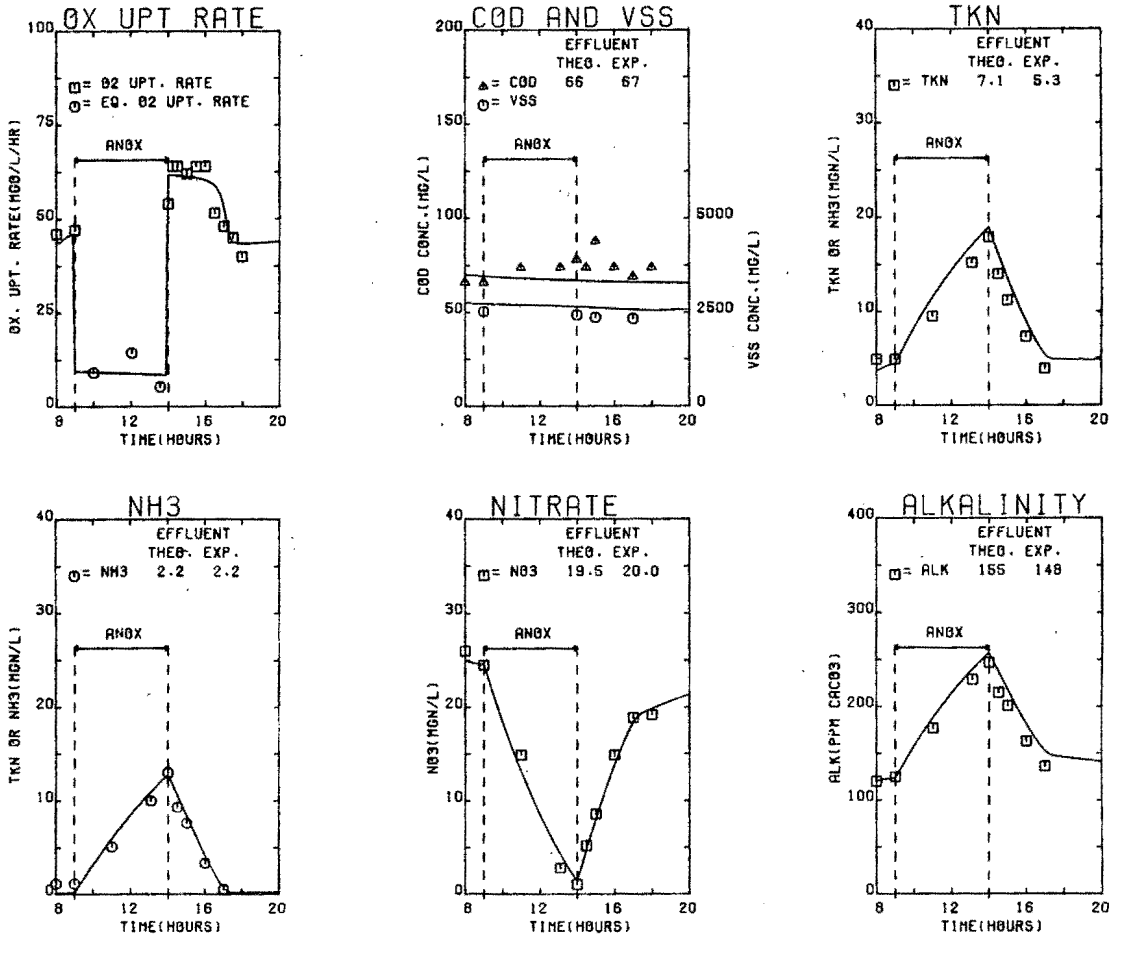

Fig.A5.11 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 5 hours per day. (Experimental data in Table A5.11)*

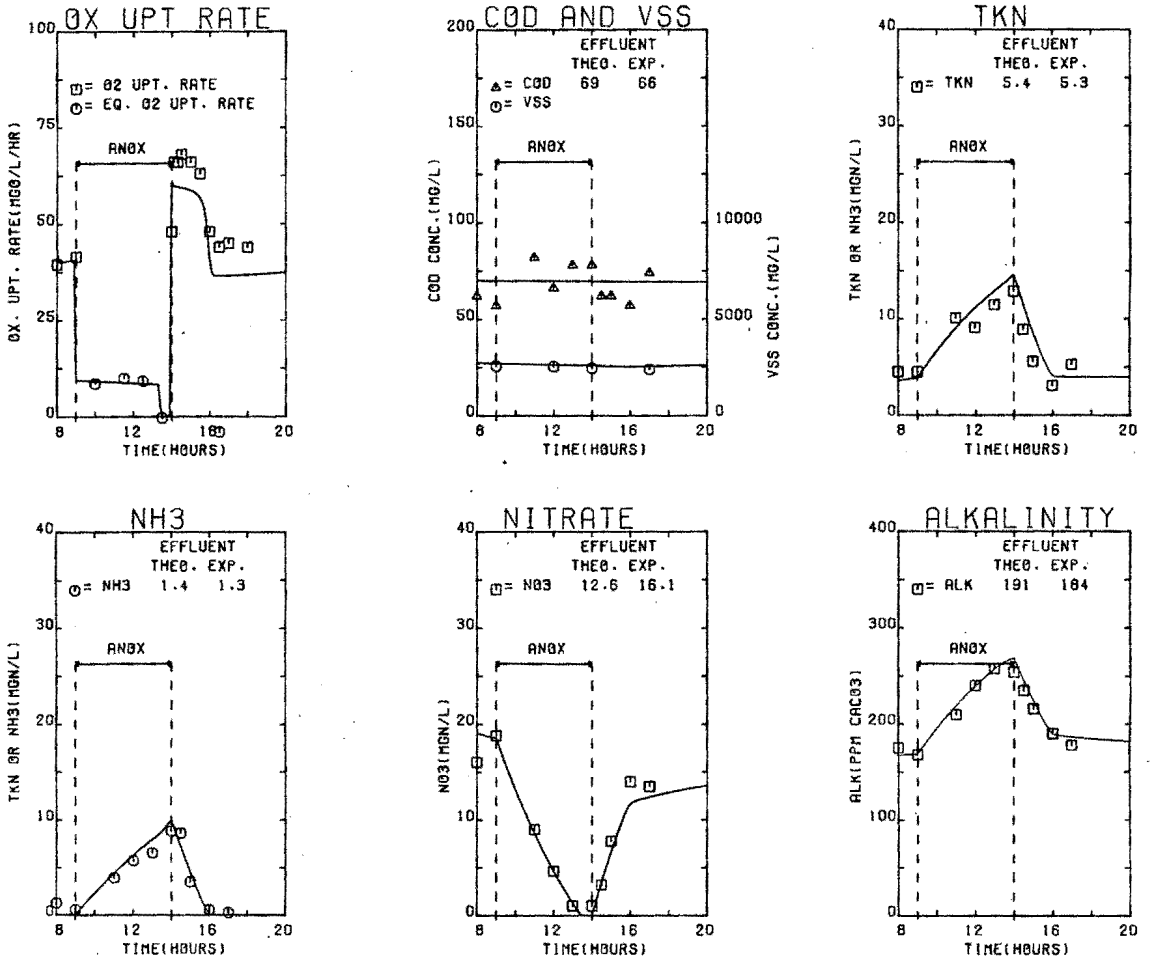

Fig.A5.12 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 5 hours per day. (Experimental data in Table A5.12)*

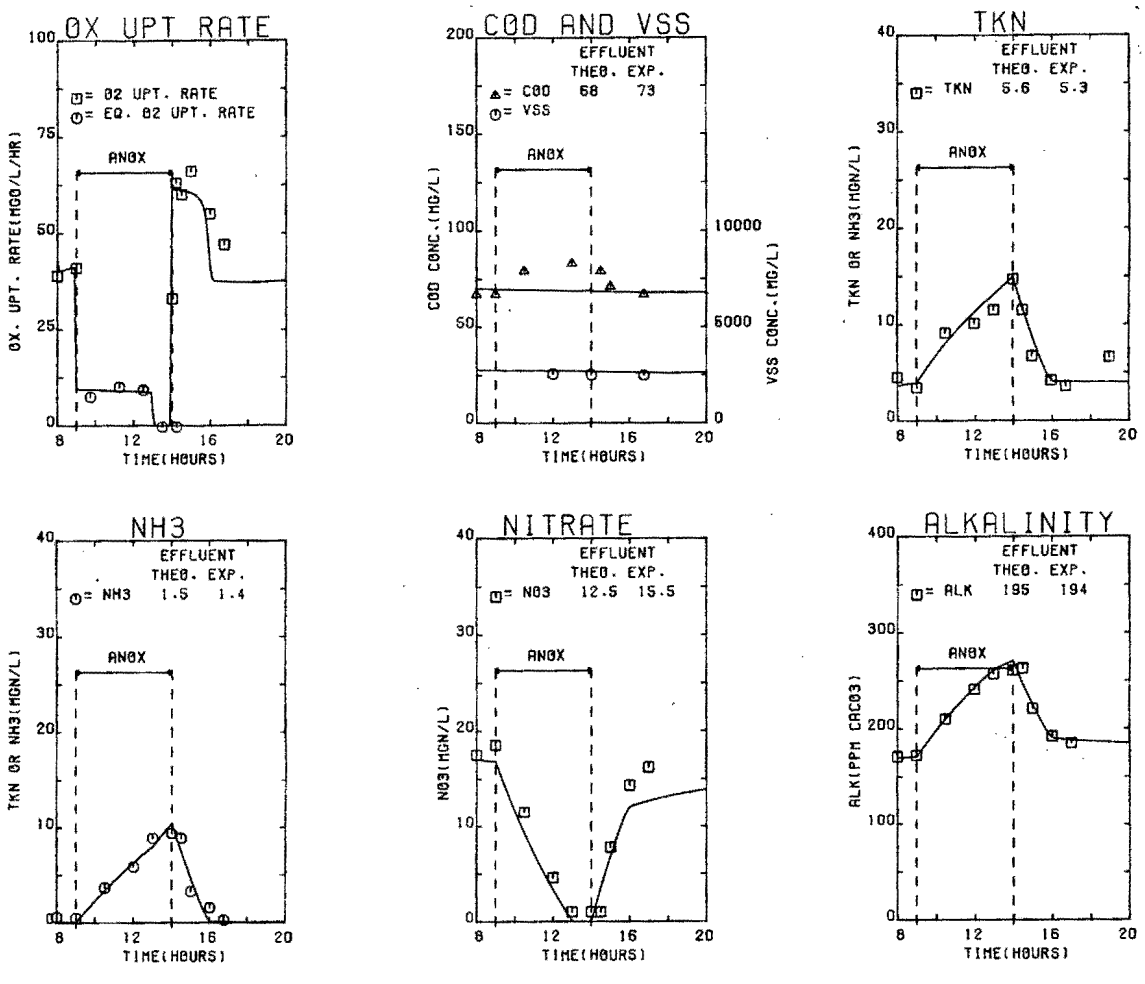

Fig.A5.13 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with an anoxic period of 5 hours per day. (Experimental data in Table A5.13)*

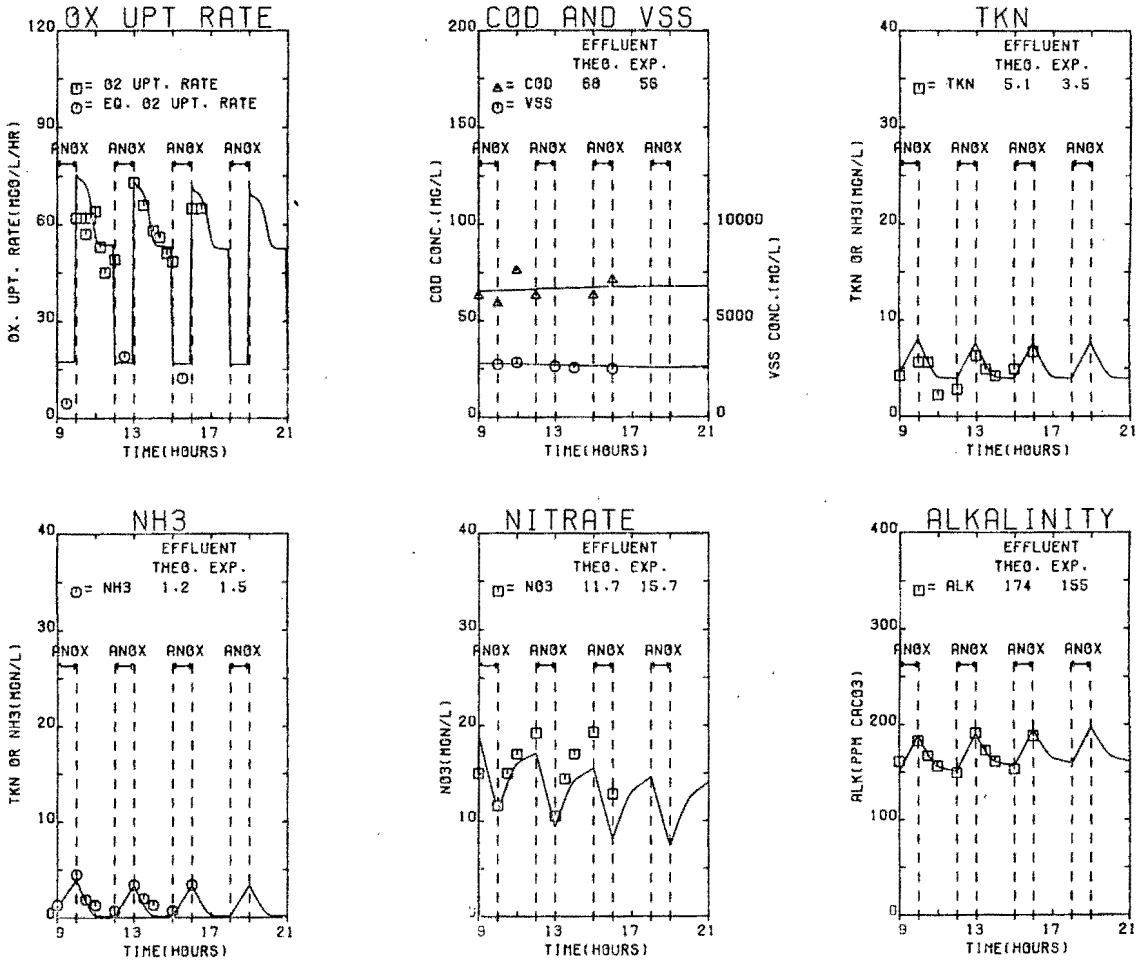

Fig.A.5.14 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with alternating one hourly anoxic and two hourly aerobic periods. (Experimental data in Table A5.15)*

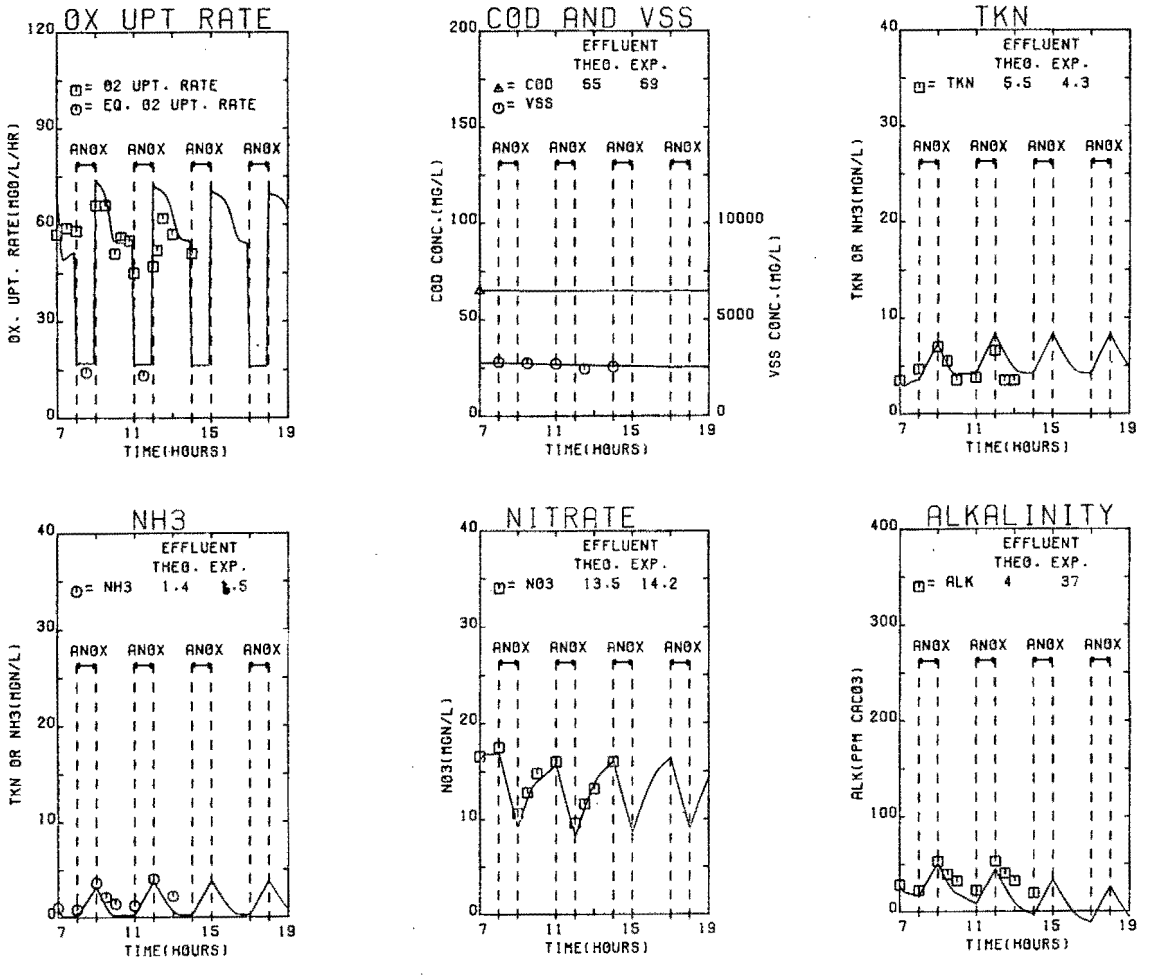

Fig.A.5.15 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with alternating one hourly anoxic and two hourly aerobic periods. (Experimental data in Table A5.14)*

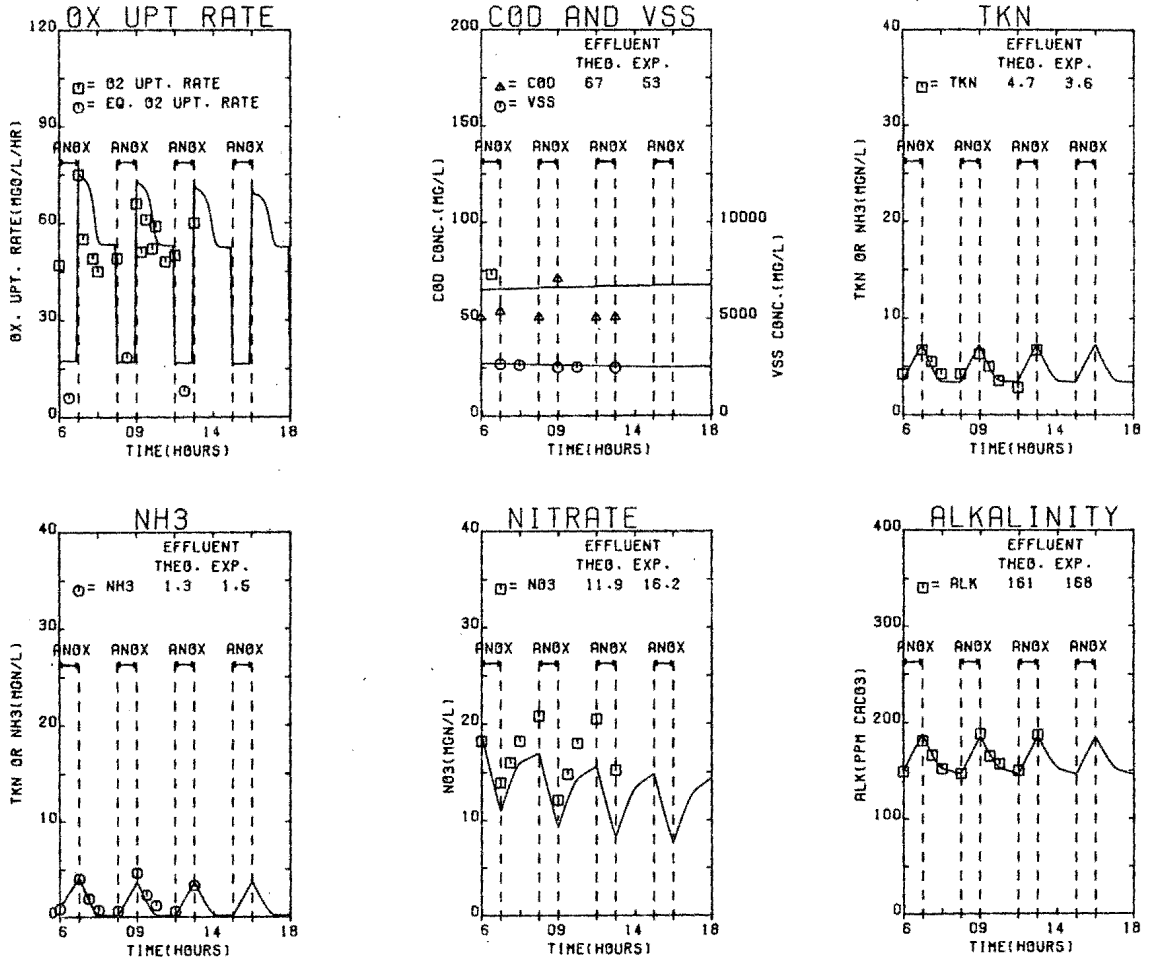

Fig. A5.16 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with alternating one hourly anoxic and two hourly aerobic periods. (Experimental data in Table A5.16)*

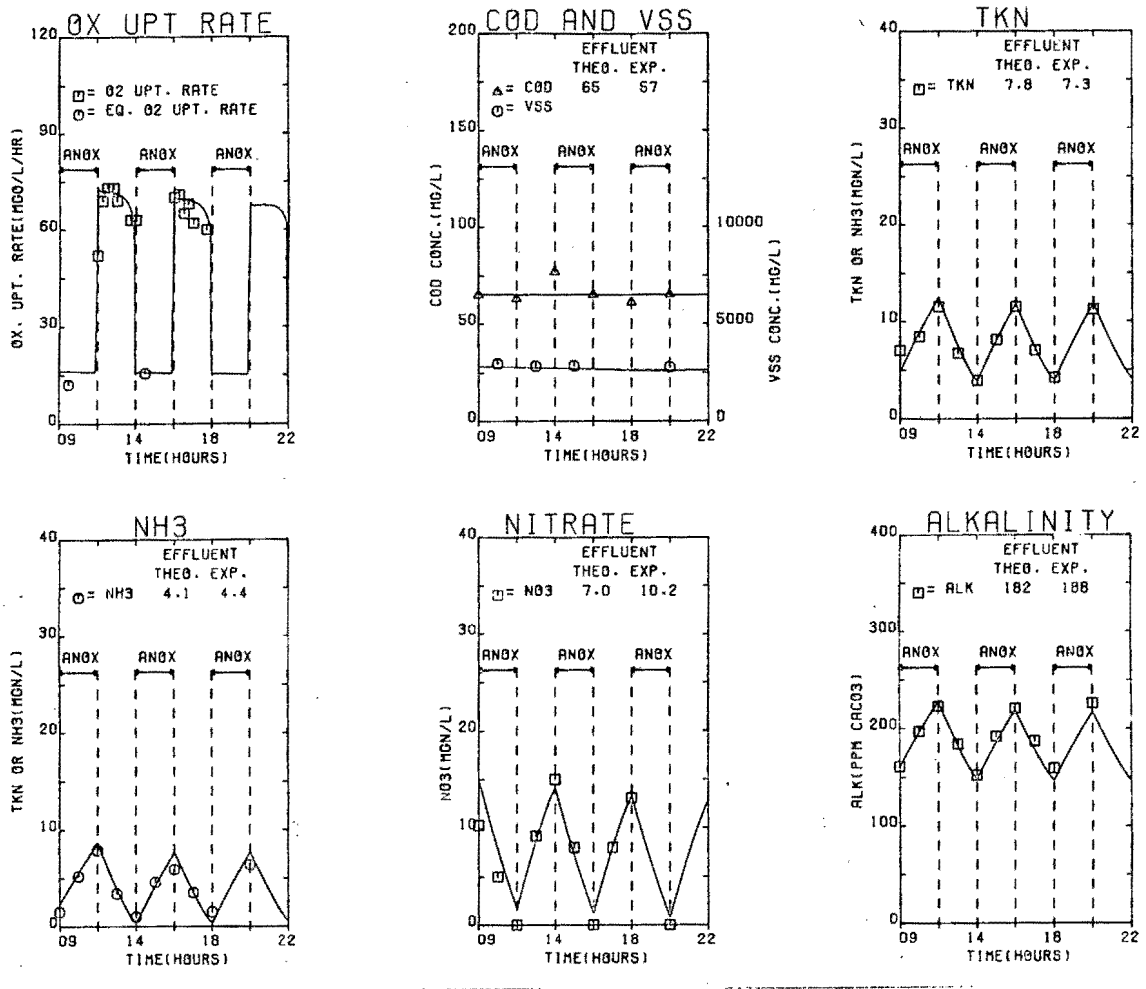

Fig.A5.17 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with alternating two hourly anoxic and two hourly aerobic periods. (Experimental data in Table A5.17)*

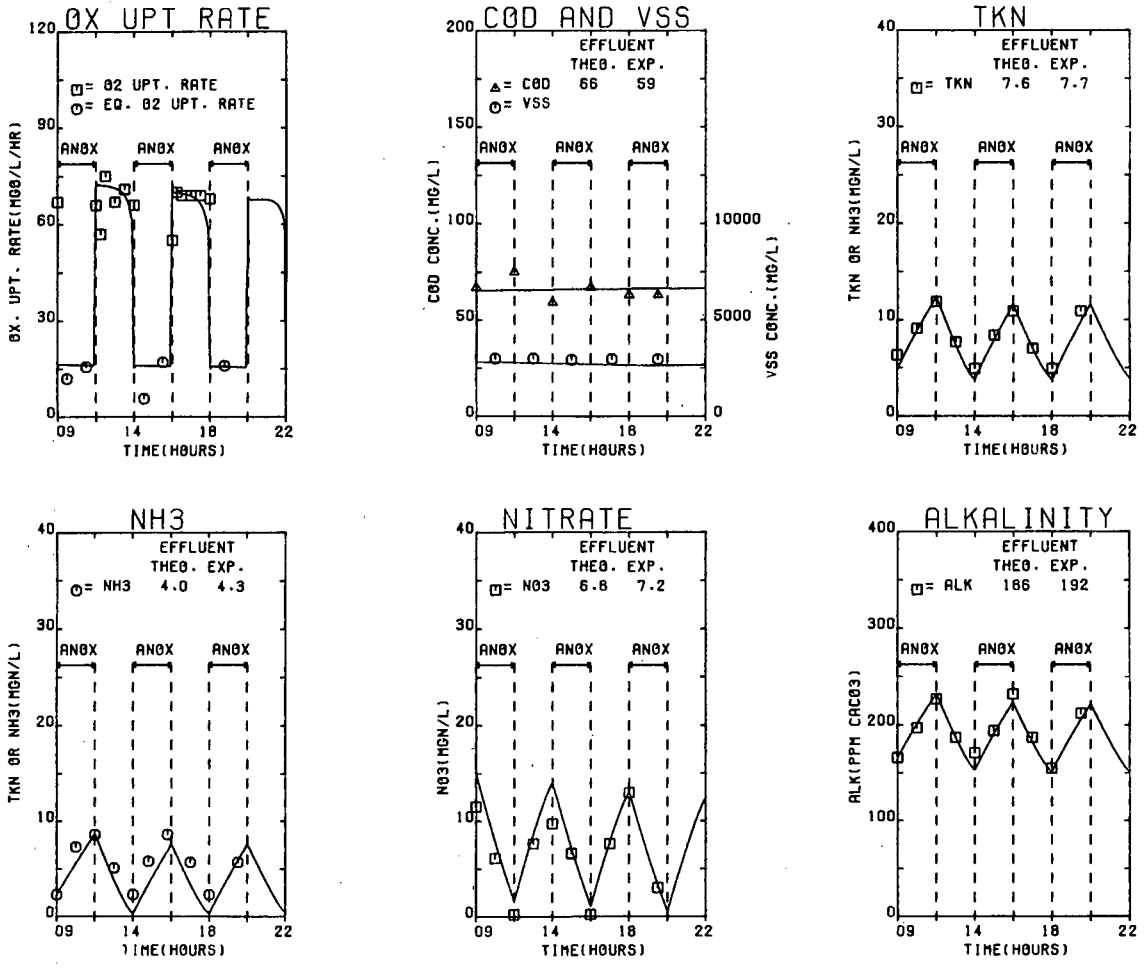

Fig. A5.18 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with alternating two hourly anoxic and two hourly aerobic periods. (Experimental data in Table A5.18)*

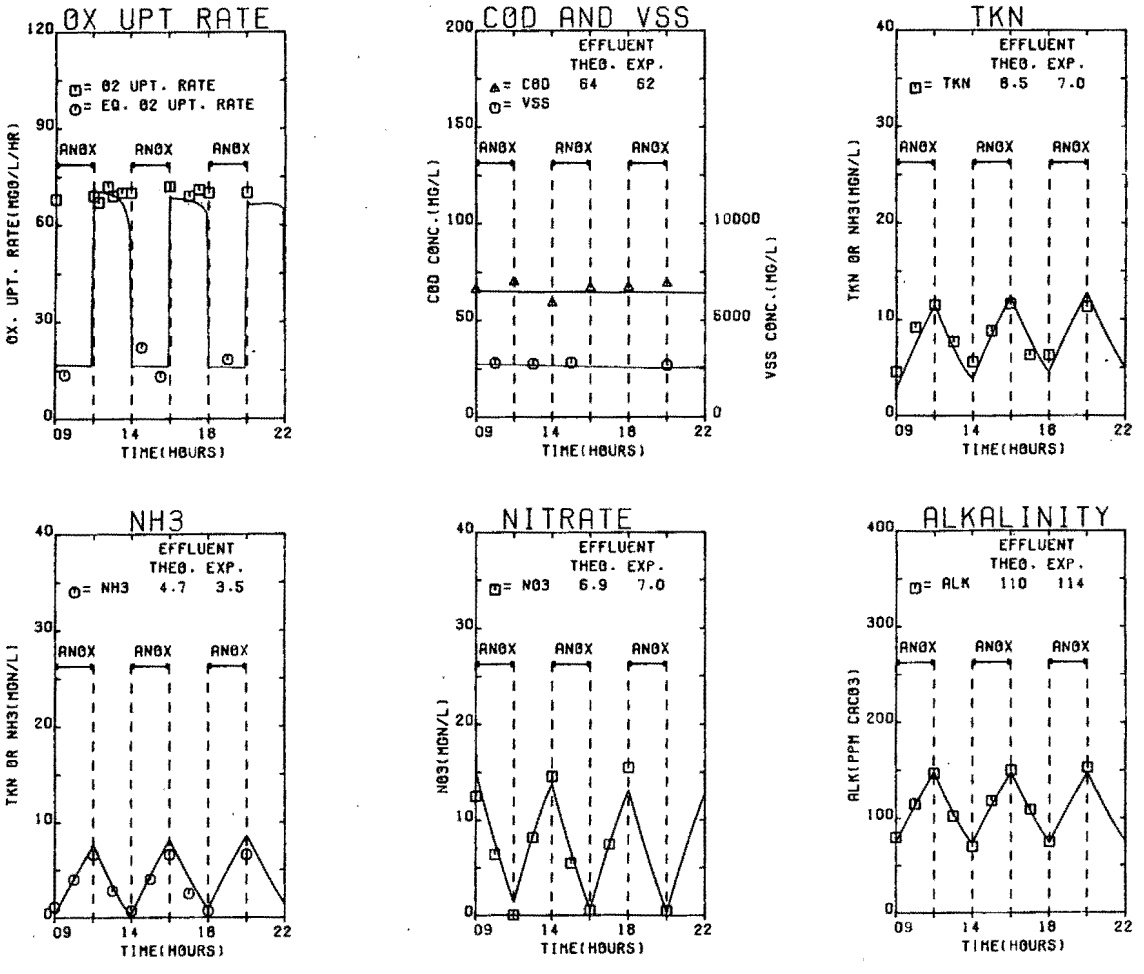

Fig.A5.19 *Experimental and simulated values of oxygen uptake rate, VSS, filtered COD, soluble TKN, Ammonia and Nitrate concentrations and alkalinity as a function of time in a single reactor system with alternating two hourly anoxic and two hourly aerobic periods. (Experimental data in Table A5.19)*

APPENDIX A6

PLOTTING PROGRAM FOR SIMULATIONS OF EXPERIMENTAL RESULTS

```
      C*************************************************************
      C       THIS PROGRAMME CAN BE USED FOR PLOTTING OF BOTH
      C       SIMULATED AND EXPERUIMENTAL DATA OF ALL THE MEASURABLE
      C       VARIABLES IN AN ACTIVATED SLUDGE PROCESS VIZ.:
      C       1) OXYGEN UPTAKE RATE (OT IN MG/L/H)
      C       2) FILETERD COD CONCENTRATION (COD IN  MG COD/L)
      C        3) VSS CONCENTRATION ( VSS IN  MG VSS/L)
      C        4) FILTERED TKN CONCENTRATION (TKN IN  MG TKN/L)
      C        5) AMMONIA CONCENTRATION (NH3-N IN  MG NH3-N/L)
      C        6) NITRATE CONCENTRATION (NO3-N IN  MG NO3-N/L)
      C        7) ALKALINITY (ALK IN MG CACO3/L)
      C       THESE PLOTS ALLOW A GRAPHICAL COMPARISON BETWEEN THE
      C       EXPERIMENTAL AND SIMULATED PROCESS RESPONSE FOR ALL
      C       THE MEASUREABLE VARIABLES.
      C
      C
      C       THE SIMULATED VALUES OF THE VARIABLES ARE READ FROM A
      C       DATA FILE GENERATED BY THE DYNAMIC STEADY STATE OR THE
      C       UNSTEADY STATE PROGRAMS. THE EXPERIMENTAL VALUES ARE
      C       ENTERED AS DATA.
      C
      C
      C       THE SUBROUTINE NAXIS IS WRITTEN BY PROF H.O.BUHR
      C       AND IS UTILIZED FOR SPECIFYING THE DEVISION OF THE
      C       Y- AND X-AXIS. THE OTHER SUBROUTINES CALLED IN THIS
      C       PROGRAM ATE PART OF THE GENERAL DISPLAY PACKAGE
      C       (G.D.P.) AVAILABLE AT THE UNIVERSITY OF CAPE TOWN.
      C*************************************************************
      C       THE NOMENCLATURE OF THE SIMULATED VARIABLES IS EXPLAINED
      C       IN THE PROGRAM FOR UNSTEADY STATE(H.UNST).
      C       FOR EXPERIMENTAL VALUES:
      C       PREFIX T REFERS TO TIME OF EXPERIMENTAL VALUE
      C       PREFIX P REFERS TO THE EXPEIMENTAL VALUE OF THE VARIABLE
      C       PREFIX N REFERS TO THE NUMBER OF EXPERIMENTAL DETERMI-
      C       NATIONS OF THE VARIABLE IN THE SET OF DATA
      C*****************************************************************
      C       NAMES BEGINNING WITH Y REFER TO THE ORDINATES OF THE DIAGRAMS
      C       YAX =LENGTH OF THE Y-AXIS
      C       YDOT=LENGTH IF INTERRUPTED LINE INDICATING ANOXIC PERIODS
      C       Y1,Y2,Y3,Y4,Y5 AND YPLUS INDICATE ORDINATES FOR TEXT
      C       YSTUK=NUMBER OF SUBDIVISIONS OF THE Y-AXIS
      C       XSTUK=NUMBER OF SUBDIVISIONS OF THE X-AXIS
      C      SIZE=SIZE OF EXPEIMENTAL          POINTS (IN MM)
      C       NDEL REFERS TO THE NUMBER OF UNITS PER CM OF ORDINATE
      C       NFIRST REFERS TO THE STARTING VALUE OF THE VARIABLES
      C       OEQ REFERS TO EQUIVALENT OXYGEN UPTAKE RATE
      C       (IE DENITRIFICATION RATE*2,86)
            PARAMETER ND=30
            PARAMETER NE=244
            INTEGER D
            REAL NT(NE),NH3(NE),NO3(NE)
            DIMENSION T(NE),OT(NE),XV(NE),ST(NE),ALK(NE),TIME(ND),TI(NE)
            DIMENSION STXV(NE),STST(NE),STNT(NE),STNH3(NE),TOEQ(ND),POEQ(ND
            DIMENSION TOT(ND),TXV(ND),TCOD(ND),TTKN(ND),TNH3(ND),TNO3(ND),
           &TALK(ND)
            DIMENSION POT(ND),PXV(ND),PCOD(ND),PTKN(ND),PNH3(ND),PNO3(ND),
           &PALK(ND)
      C     READ IN SIMULATED DATA
            READ(18,35)ECOD,ETKN,ENH3,ENO3,EALK
            WRITE(5,35)ECOD,ETKN,ENH3,ENO3,EALK
            READ(18,35)KOM,D,INIT,ANOX,AROB,IFREQ,NDH
            WRITE(5,35)KOM,D,INIT,ANOX,AROB,IFREQ,NDH
            NFIRST=NDH+1
```

```
      NDEL=NDH+2
      READ(18,45)(OT(I),XV(I),ST(I),NT(I),NH3(I),NO3(I),ALK(I),I=1,NDH
      READ(18,35)START,END
      WRITE(5,35)START,END
      DO 36 I=1,2501,100
      II=I-1
      IF((KOM-II).LT.100)GO TO 37
36    CONTINUE
37    T(1)=II/100.0+(KOM-I1)/60.0
      READ(8,15)NOT,NXV,NCOD,NTKN,NNH3,NNO3,NALK
      READ(8,15)AVCOD,AVTKN,AVNH3,AVNO3,AVALK
15    FORMAT()
      READ(8,15)(TOT(I),POT(I),I=1,NOT)
      READ(8,15)(TXV(I),PXV(I),I=1,NXV)
      READ(8,15)(TCOD(I),PCOD(I),I=1,NCOD)
      READ(8,15)(TTKN(I),PTKN(I),I=1,NTKN)
      READ(8,15)(TNH3(I),PNH3(I),I=1,NNH3)
      READ(8,15)(TNO3(I),PNO3(I),I=1,NNO3)
      READ(8,15)(TALK(I),PALK(I),I=1,NALK)
      READ(8,15)YAX,YDOT,Y0,Y1,Y2,Y3,Y4,Y5,YPLUS,KJ,YFEED,YWRITE,NAMBA
      Y5=Y5+0.5
      READ(8,15)T(NDEL),OT(NDEL),ST(NDEL),NT(NDEL),NH3(NDEL),
     &NO3(NDEL),ALK(NDEL),XV(NDEL)
      READ(8,15)XSTUK,YSTUK,ITYD,IJ,SIZE,TRN
      NN=(NDH-1)/10
      XAX=FLOAT(NN)
      XVSS=XAX+6*SIZE
      DO 87 I=1,50,1
      IF(I.GT.NOT) GO TO 887
      IF(TOT(I).LT.T(1)) TOT(I)=TOT(I)+24.0
887   IF(I.GT.NXV) GO TO 888
      IF(TXV(I).LT.T(1)) TXV(I)=TXV(I)+24.0
888   IF(I.GT.NCOD) GO TO 889
      IF(TCOD(I).LT.T(1)) TCOD(I)=TCOD(I)+24.0
889   IF(I.GT.NTKN) GO TO 890
      IF(TTKN(I).LT.T(1)) TTKN(I)=TTKN(I)+24.0
890   IF(I.GT.NNH3) GO TO 891
      IF(TNH3(I).LT.T(1)) TNH3(I)=TNH3(I)+24.0
891   IF(I.GT.NNO3) GO TO 892
      IF(TNO3(I).LT.T(1)) TNO3(I)=TNO3(I)+24.0
892   IF(I.GT.NALK) GO TO 87
      IF(TALK(I).LT.T(1)) TALK(I)=TALK(I)+24.0
87    CONTINUE
      J=1
      DO 3 I=1,NNO3,1
      IF(PNO3(I+1).GT.PNO3(I)) GO TO 3
      IF(PNO3(I+1).EQ.0.0)GO TO 3
      IF( TNO3(I+1).LE.(INIT+T(1))) GO TO 3
      RNO3=(PNO3(I)-PNO3(I+1))/(TNO3(I+1)-TNO3(I))
     &-(PNO3(I)+PNO3(I+1))/(2.0*TRN)
      POEQ(J)=2.86*RNO3
      TOEQ(J)=(TNO3(I+1)+TNO3(I))/2.0
      J=J+1
3     CONTINUE
      K=J
      T(NFIRST)=T(1)
      OT(NFIRST)=0.0
      XV(NFIRST)=0.0
      ST(NFIRST)=0.0
      NT(NFIRST)=0.0
      NH3(NFIRST)=0.0
      NO3(NFIRST)=0.0
      ALK(NFIRST)=0.0
      DO 65 I=1,NOT,1
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```
128             TOT(I)=(TOT(I)-T(NFIRST))/T(NDEL)
129             POT(I)=(POT(I)-OT(NFIRST))/OT(NDEL)
130       65    CONTINUE
131             DO 56 I=1,K,1
132             TOEQ(I)=(TOEQ(I)-T(NFIRST))/T(NDEL)
133             POEQ(I)=(POEQ(I)-OT(NFIRST))/OT(NDEL)
134       56    CONTINUE
135             DO 75 I=1,NXV,1
136             TXV(I)=(TXV(I)-T(NFIRST))/T(NDEL)
137             PXV(I)=(PXV(I)-XV(NFIRST))/XV(NDEL)
138       75    CONTINUE
139             DO 85 I=1,NCOD,1
140             TCOD(I)=(TCOD(I)-T(NFIRST))/T(NDEL)
141             PCOD(I)=(PCOD(I)-ST(NFIRST))/ST(NDEL)
142       85    CONTINUE
143             DO 95 I=1,NTKN,1
144             TTKN(I)=(TTKN(I)-T(NFIRST))/T(NDEL)
145             PTKN(I)=(PTKN(I)-NT(NFIRST))/NT(NDEL)
146       95    CONTINUE
147             DO 105 I=1,NNH3,1
148             TNH3(I)=(TNH3(I)-T(NFIRST))/T(NDEL)
149             PNH3(I)=(PNH3(I)-NH3(NFIRST))/NH3(NDEL)
150       105   CONTINUE
151             DO 115 I=1,NNO3,1
152             TNO3(I)=(TNO3(I)-T(NFIRST))/T(NDEL)
153             PNO3(I)=(PNO3(I)-NO3(NFIRST))/NO3(NDEL)
154       115   CONTINUE
155             DO 125 I=1,NALK,1
156             TALK(I)=(TALK(I)-T(NFIRST))/T(NDEL)
157             PALK(I)=(PALK(I)-ALK(NFIRST))/ALK(NDEL)
158       125   CONTINUE
159             DO 38 I=2,NDH,1
160             T(I)=(T(I-1)+24.0/D)
161       38    CONTINUE
162             DO 335 I=1,NDH,1
163             TI(I)=(T(I)-T(1))/T(NDEL)
164             STXV(I)=(XV(I)-XV(NFIRST))/XV(NDEL)
165             STST(I)=(ST(I)-ST(NFIRST))/ST(NDEL)
166             STNT(I)=(NT(I)-NT(NFIRST))/NT(NDEL)
167             STNH3(I)=(NH3(I)-NH3(NFIRST))/NH3(NDEL)
168       335   CONTINUE
169             TIME(1)=INIT/T(NDEL)
170             DO 205 I=1,IFREQ,1
171             J=2*I-1
172             TIME(J+1)=TIME(J)+ANOX/T(NDEL)
173             TIME(J+2)=TIME(J+1)+AROB/T(NDEL)
174       205   CONTINUE
175             START=START/T(NDEL)+SIZE
176             END=END/T(NDEL)
177             NFR=IFREQ+1
178       45    FORMAT(7E13.5)
179       35    FORMAT()
180             X2=XAX+10.0
181             XPAGE=3.0*X2
182             X3=-2.0*X2
183             CALL PLOTS(0,0,0)
184             CALL FACTOR (0.33)
185             CALL NEWPEN(1)
186             CALL OPMES(24,'PLEASE LOAD P1-BK/I4    ')
187             CALL PAGDEF (-5.0,-6.0,XPAGE,52.0)
188             CALL PLOT(0.0,26.0,-3)
189             CALL NAXIS(0.0,0.0,XAX,0,XSTUK,0.0,1,0.25,2)
190             CALL NAXIS(0.0,YAX,XAX,0,XSTUK,0.0,-1,0.25,2)
191             CALL NAXIS(0.0,0.0,YAX,0,YSTUK,90.0,-1,0.25,2)
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192             CALL NAXIS(XAX,0.0,YAX,0,YSTUK,90.0,1,0.25,2)
193             DO 106 I=1,ITYD,1
194             TYD=T(1)+(I-1)*XAX*T(NDEL)/(ITYD-1)
195             IF(TYD.GT.24.0)TYD=TYD-24.0
196             XVAL=(I-1)*XAX/(ITYD-1.0)-0.5
197             IF(XVAL.LT.0.0) XVAL=0.0
198             CALL NUMBER(XVAL,-1.0,SIZE,TYD,0.0,-1)
199       106   CONTINUE
200             DO 101 I=1,IJ,1
201             YOT=(I-1)*YAX*OT(NDEL)/(IJ-1)
202             YVAL=YOT/OT(NDEL)
203             IF(YOT.LT.9.9) GO TO 301
204             IF(YOT.GT.9.9.AND.YOT.LT.99.9)GO TO 302
205             IF(YOT.GT.99.9)GO TO 303
206       301   CALL NUMBER(-0.5,YVAL,SIZE,YOT,0.0,-1)
207             GO TO 101
208       302   CALL NUMBER(-1.0,YVAL,SIZE,YOT,0.0,-1)
209             GO TO 101
210       303   CALL NUMBER(-1.5,YVAL,SIZE,YOT,0.0,-1)
211       101   CONTINUE
212             CALL SYMBOL(3.5,-2.0,SIZE,'TIME(HOURS)',0.0,11)
213             CALL SYMBOL(-2.0,4.0,SIZE,'OX. UPT. RATE(MGO/L/HR)',90.0,23)
214             DO 215 I=1,NFR,1
215             DO 215 J=1,KJ,1
216             Y=J-1.0
217             IF(TIME(I).GT.XAX) GO TO 215
218             CALL SYMBOL(TIME(I),Y,0.5,13,0.0,-1)
219       215   CONTINUE
220             IF(NAMBA.EQ.1) GO TO 256
221             TEXT=START +2.5
222             CALL SYMBOL(START,YFEED,SIZE,22,90.0,-1)
223             CALL SYMBOL(END,YFEED,SIZE,23,90.0,-2)
224             CALL SYMBOL(TEXT,YWRITE,SIZE,'FEEDING PERIOD',0.0,14)
225       256   CONTINUE
226             DO 255 I=1,IFREQ,2
227             BEGIN=TIME(I)+SIZE
228             CALL SYMBOL(BEGIN,YDOT,SIZE,22,90.0,-1)
229             CALL SYMBOL(TIME(I+1),YDOT,SIZE,23,90.0,-2)
230             X=(TIME(I)+TIME(I+1))/2-0.75
231             CALL SYMBOL(X,Y5,SIZE,'ANOX',0.0,4)
232       255   CONTINUE
233             CALL SYMBOL(0.5,YPLUS,1.0,'OX UPT RATE',0.0,11)
234             CALL SYMBOL(1.0,Y2,SIZE,0,0.0,-1)
235             CALL SYMBOL(1.0,Y4,SIZE,1,0.0,-1)
236             CALL SYMBOL(1.5,Y2,SIZE,'= O2 UPT. RATE',0.0,14)
237             CALL SYMBOL(1.5,Y4,SIZE,'= EQ. O2 UPT. RATE',0.0,18)
238             DO 55 I=1,NOT,1
239             CALL SYMBOL(TOT(I),POT(I),SIZE,0,0.0,-1)
240        55    CONTINUE
241             DO 44 I=1,K,1
242             CALL SYMBOL(TOEQ(I),POEQ(I),SIZE,1,0.0,-1)
243        44    CONTINUE
244             CALL LINE(T,OT,NDH,1,0,0)
245       16    FORMAT(1H ,5X,'T=',F13.5,5X,'OT=',F13.5)
246             CALL PLOT (X2,0.0,-3)
247             CALL NAXIS(0.0,0.0,XAX,0,XSTUK,0.0,1,0.25,2)
248             CALL NAXIS(0.0,YAX,XAX,0,XSTUK,0.0,-1,0.25,2)
249             CALL NAXIS(0.0,0.0,YAX,0,YSTUK,90.0,-1,0.25,2)
250             CALL NAXIS(XAX,0.0,YAX,0,YSTUK,90.0,1,0.25,2)
251             DO 107 I=1,ITYD,1
252             TYD=T(1)+(I-1)*XAX*T(NDEL)/(ITYD-1)
253             IF(TYD.GT.24.0)TYD=TYD-24.0
254             XVAL=(I-1)*XAX/(ITYD-1.0)-0.5
255             IF(XVAL.LT.0.0) XVAL=0.0
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256             CALL NUMBER(XVAL,-1.0,SIZE,TYD,0.0,-1)
257       107   CONTINUE
258             IF(NAMBA.EQ.1) GO TO 257
259             CALL SYMBOL(START,YFEED,SIZE,22,90.0,-1)
260             CALL SYMBOL(END,YFEED,SIZE,23,90.0,-2)
261             CALL SYMBOL(TEXT,YWRITE,SIZE,'FEEDING PERIOD',0.0,14)
262       257   CONTINUE
263             DO 102 I=1,IJ,1
264             YST=(I-1)*YAX*ST(NDEL)/(IJ-1)
265             YVAL=YST/ST(NDEL)
266             IF(YST.LT.9.9) GO TO 401
267             IF(YST.GT.9.9.AND.YST.LT.99.9)GO TO 402
268             IF(YST.GT.99.9)GO TO 403
269       401   CALL NUMBER(-0.5,YVAL,SIZE,YST,0.0,-1)
270             GO TO 102
271       402   CALL NUMBER(-1.0,YVAL,SIZE,YST,0.0,-1)
272             GO TO 102
273       403   CALL NUMBER(-1.5,YVAL,SIZE,YST,0.0,-1)
274       102   CONTINUE
275             XAX1=XAX+SIZE
276             DO 111 I=1,3,1
277             YXV=(I-1)*YAX*XV(NDEL)/(IJ-1)
278             YVAL=YXV/XV(NDEL)
279             IF(YXV.LT.9.9) GO TO 501
280             IF(YXV.GT.9.9.AND.YXV.LT.99.9)GO TO 502
281             IF(YXV.GT.99.9)GO TO 503
282       501   CALL NUMBER(XAX1,YVAL,SIZE,YXV,0.0,-1)
283             GO TO 111
284       502   CALL NUMBER(XAX1,YVAL,SIZE,YXV,0.0,-1)
285             GO TO 111
286       503   CALL NUMBER(XAX1,YVAL,SIZE,YXV,0.0,-1)
287       111   CONTINUE
288             CALL SYMBOL(3.5,-2.0,SIZE,'TIME(HOURS)',0.0,11)
289             CALL SYMBOL (-2.0,6.0,SIZE,'COD CONC.(MG/L)',90.0,15)
290             CALL SYMBOL (XVSS,2.0,SIZE,'VSS CONC.(MG/L)',90.0,15)
291             DO 315 I=1,NFR,1
292             DO 315 J=1,KJ,1
293             Y=J-1.0
294             IF(TIME(I).GT.XAX) GO TO 315
295             CALL SYMBOL(TIME(I),Y,0.5,13,0.0,-1)
296       315   CONTINUE
297             DO 355 I=1,IFREQ,2
298             BEGIN=TIME(I)+SIZE
299             CALL SYMBOL(BEGIN,YDOT,SIZE,22,90.0,-1)
300             CALL SYMBOL(TIME(I+1),YDOT,SIZE,23,90.0,-2)
301             X=(TIME(I)+TIME(I+1))/2-0.75
302             CALL SYMBOL(X,Y5,SIZE,'ANOX',0.0,4)
303       355   CONTINUE
304             CALL SYMBOL(0.5,YPLUS,1.0,'COD AND VSS',0.0,11)
305             CALL SYMBOL(1.0,Y4,SIZE,1,0.0,-1)
306             CALL SYMBOL(1.0,Y2,SIZE,2,0.0,-1)
307             CALL SYMBOL(1.5,Y4,SIZE,'= VSS',0.0,5)
308             CALL SYMBOL(1.5,Y2,SIZE,'= COD',0.0,5)
309             CALL SYMBOL(5.5,Y0,SIZE,'EFFLUENT',0.0,8)
310             CALL SYMBOL(5.0,Y1,SIZE,'THEO. EXP.',0.0,10)
311             CALL NUMBER(5.5,Y2,SIZE,ECOD,0.0,-1)
312             CALL NUMBER(8.5,Y2,SIZE,AVCOD,0.0,-1)
313             DO 135 I=1,NXV,1
314             CALL SYMBOL(TXV(I),PXV(I),SIZE,1,0.0,-1)
315       135   CONTINUE
316             DO 145 I=1,NCOD,1
317             CALL SYMBOL(TCOD(I),PCOD(I),SIZE,2,0.0,-1)
318       145   CONTINUE
319             CALL LINE(T,XV,NDH,1,0,0)
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320           CALL LINE(T,ST,NDH,1,0,0)
321           CALL PLOT (X2,0.0,-3)
322           CALL NAXIS(0.0,0.0,XAX,0,XSTUK,0.0,1,0.25,2)
323           CALL NAXIS(0.0,YAX,XAX,0,XSTUK,0.0,-1,0.25,2)
324           CALL NAXIS(0.0,0.0,YAX,0,YSTUK,90.0,-1,0.25,2)
325           CALL NAXIS(XAX,0.0,YAX,0,YSTUK,90.0,1,0.25,2)
326           DO 109 I=1,ITYD,1
327           TYD=T(1)+(I-1)*XAX*T(NDEL)/(ITYD-1)
328           IF(TYD.GT.24.0)TYD=TYD-24.0
329           XVAL=(I-1)*XAX/(ITYD-1.0)-0.5
330           IF(XVAL.LT.0.0) XVAL=0.0
331           CALL NUMBER(XVAL,-1.0,SIZE,TYD,0.0,-1)
332       109 CONTINUE
333           IF(NAMBA.EQ.1) GO TO 258
334           CALL SYMBOL(START,YFEED,SIZE,22,90.0,-1)
335           CALL SYMBOL(END,YFEED,SIZE,23,90.0,-2)
336           CALL SYMBOL(TEXT,YWRITE,SIZE,'FEEDING PERIOD',0.0,14)
337       258 CONTINUE
338           DO 104 I=1,IJ,1
339           YNH3=(I-1)*YAX*NH3(NDEL)/(IJ-1)
340           YVAL=YNH3/NH3(NDEL)
341           IF(YNH3.LT.9.9) GO TO 601
342           IF(YNH3.GT.9.9.AND.YNH3.LT.99.9)GO TO 602
343           IF(YNH3.GT.99.9)GO TO 603
344       601 CALL NUMBER(-0.5,YVAL,SIZE,YNH3,0.0,-1)
345           GO TO 104
346       602 CALL NUMBER(-1.0,YVAL,SIZE,YNH3,0.0,-1)
347           GO TO 104
348       603 CALL NUMBER(-1.5,YVAL,SIZE,YNH3,0.0,-1)
349       104 CONTINUE
350           CALL SYMBOL(3.5,-2.0,SIZE,'TIME(HOURS)',0.0,11)
351           CALL SYMBOL(-2.0,6.0,SIZE,'TKN OR NH3(MGN/L)',90.0,17)
352           DO 225 I=1,NFR,1
353           DO 225 J=1,KJ,1
354           Y=J-1.0
355           IF(TIME(I).GT.XAX) GO TO 225
356           CALL SYMBOL(TIME(I),Y,0.5,13,0.0,-1)
357       225 CONTINUE
358           DO 265 I=1,IFREQ,2
359           BEGIN=TIME(I)+SIZE
360           CALL SYMBOL(BEGIN,YDOT,SIZE,22,90.0,-1)
361           CALL SYMBOL(TIME(I+1),YDOT,SIZE,23,90.0,-2)
362           X=(TIME(I)+TIME(I+1))/2-0.75
363           CALL SYMBOL(X,Y5,SIZE,'ANOX',0.0,4)
364       265 CONTINUE
365           CALL SYMBOL(4.0,YPLUS,1.0,'TKN',0.0,3)
366           CALL SYMBOL(1.0,Y2,SIZE,0,0.0,-1)
367           CALL SYMBOL(1.5,Y2,SIZE,'= TKN',0.0,5)
368           CALL SYMBOL(5.5,Y0,SIZE,'EFFLUENT',0.0,8)
369           CALL SYMBOL(5.0,Y1,SIZE,'THEO. EXP.',0.0,10)
370           CALL NUMBER(5.5,Y2,SIZE,ETKN,0.0,1)
371           CALL NUMBER(8.5,Y2,SIZE,AVTKN,0.0,1)
372           DO 155 I=1,NTKN,1
373           CALL SYMBOL(TTKN(I),PTKN(I),SIZE,0,0.0,-1)
374        155 CONTINUE
375           CALL LINE(T,NT,NDH,1,0,0)
376           CALL PLOT (X3,-26.0,-3)
377           CALL NAXIS(0.0,0.0,XAX,0,XSTUK,0.0,1,0.25,2)
378           CALL NAXIS(0.0,YAX,XAX,0,XSTUK,0.0,-1,0.25,2)
379           CALL NAXIS(0.0,0.0,YAX,0,YSTUK,90.0,-1,0.25,2)
380           CALL NAXIS(XAX,0.0,YAX,0,YSTUK,90.0,1,0.25,2)
381           DO 114 I=1,ITYD,1
382           TYD=T(1)+(I-1)*XAX*T(NDEL)/(ITYD-1)
383           IF(TYD.GT.24.0)TYD=TYD-24.0
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384             XVAL=(I-1)*XAX/(ITYD-1.0)-0.5
385             IF(XVAL.LT.0.0) XVAL=0.0
386             CALL NUMBER(XVAL,-1.0,SIZE,TYD,0.0,-1)
387         114 CONTINUE
388             DO 116 I=1,IJ,1
389             YNH3=(I-1)*YAX*NH3(NDEL)/(IJ-1)
390             YVAL=YNH3/NH3(NDEL)
391             IF(YNH3.LT.9.9) GO TO 901
392             IF(YNH3.GT.9.9.AND.YNH3.LT.99.9)GO TO 902
393             IF(YNH3.GT.99.9)GO TO 903
394         901 CALL NUMBER(-0.5,YVAL,SIZE,YNH3,0.0,-1)
395             GO TO 116
396         902 CALL NUMBER(-1.0,YVAL,SIZE,YNH3,0.0,-1)
397             GO TO 116
398         903 CALL NUMBER(-1.5,YVAL,SIZE,YNH3,0.0,-1)
399         116 CONTINUE
400             CALL SYMBOL(3.5,-2.0,SIZE,'TIME(HOURS)',0.0,11)
401             CALL SYMBOL(-2.0,6.0,SIZE,'TKN OR NH3(MGN/L)',90.0,17)
402             DO 226 I=1,NFR,1
403             DO 226 J=1,KJ,1
404             Y=J-1.0
405             IF(TIME(I).GT.XAX) GO TO 226
406             CALL SYMBOL(TIME(I),Y,0.5,13,0.0,-1)
407         226 CONTINUE
408             DO 266 I=1,IFREQ,2
409             BEGIN=TIME(I)+SIZE
410             CALL SYMBOL(BEGIN,YDOT,SIZE,22,90.0,-1)
411             CALL SYMBOL(TIME(I+1),YDOT,SIZE,23,90.0,-2)
412             X=(TIME(I)+TIME(I+1))/2-0.75
413             CALL SYMBOL(X,Y5,SIZE,'ANOX',0.0,4)
414         266 CONTINUE
415             CALL SYMBOL(4.0,YPLUS,1.0,'NH3',0.0,3)
416             CALL SYMBOL(5.5,Y0,SIZE,'EFFLUENT',0.0,8)
417             CALL SYMBOL(5.0,Y1,SIZE,'THEO. EXP.',0.0,10)
418             CALL SYMBOL(1.0,Y2,SIZE,1,0.0,-1)
419             CALL SYMBOL(1.5,Y2,SIZE,'= NH3',0.0,5)
420             CALL NUMBER(5.5,Y2,SIZE,ENH3,0.0,1)
421             CALL NUMBER(8.5,Y2,SIZE,AVNH3,0.0,1)
422             DO 165 I=1,NNH3,1
423             CALL SYMBOL(TNH3(I),PNH3(I),SIZE,1,0.0,-1)
424         165 CONTINUE
425             IF(NAMBA.EQ.1) GO TO 259
426             CALL SYMBOL(START,YFEED,SIZE,22,90.0,-1)
427             CALL SYMBOL(END,YFEED,SIZE,23,90.0,-2)
428             CALL SYMBOL(TEXT,YWRITE,SIZE,'FEEDING PERIOD',0.0,14)
429         259 CONTINUE
430             CALL LINE(T,NH3,NDH,1,0,0)
431             CALL PLOT (X2,0.0,-3)
432             CALL NAXIS(0.0,0.0,XAX,0,XSTUK,0.0,1,0.25,2)
433             CALL NAXIS(0.0,YAX,XAX,0,XSTUK,0.0,-1,0.25,2)
434             CALL NAXIS(0.0,0.0,YAX,0,YSTUK,90.0,-1,0.25,2)
435             CALL NAXIS(XAX,0.0,YAX,0,YSTUK,90.0,1,0.25,2)
436             DO 112 I=1,ITYD,1
437             TYD=T(1)+(I-1)*XAX*T(NDEL)/(ITYD-1)
438             IF(TYD.GT.24.0)TYD=TYD-24.0
439             XVAL=(I-1)*XAX/(ITYD-1.0)-0.5
440             IF(XVAL.LT.0.0) XVAL=0.0
441             CALL NUMBER(XVAL,-1.0,SIZE,TYD,0.0,-1)
442         112 CONTINUE
443             DO 113 I=1,IJ,1
444             YNO3=(I-1)*YAX*NO3(NDEL)/(IJ-1)
445             YVAL=YNO3/NO3(NDEL)
446             IF(YNO3.LT.9.9) GO TO 801
447             IF(YNO3.GT.9.9.AND.YNO3.LT.99.9)GO TO 802
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448             IF(YNO3.GT.99.9)GO TO 803
449       801   CALL NUMBER(-0.5,YVAL,SIZE,YNO3,0.0,-1)
450             GO TO 113
451       802   CALL NUMBER(-1.0,YVAL,SIZE,YNO3,0.0,-1)
452             GO TO 113
453       803   CALL NUMBER(-1.5,YVAL,SIZE,YNO3,0.0,-1)
454       113   CONTINUE
455             CALL SYMBOL(3.5,-2.0,SIZE,'TIME(HOURS)',0.0,11)
456             CALL SYMBOL(-1.5,6.5,SIZE,'NO3(MGN/L)',90.0,10)
457             DO 425 I=1,NFR,1
458             DO 425 J=1,KJ,1
459             Y=J-1.0
460             IF(TIME(I).GT.XAX) GO TO 425
461             CALL SYMBOL(TIME(I),Y,0.5,13,0.0,-1)
462       425   CONTINUE
463             DO 465 I=1,IFREQ,2
464             BEGIN=TIME(I)+SIZE
465             CALL SYMBOL(BEGIN,YDOT,SIZE,22,90.0,-1)
466             CALL SYMBOL(TIME(I+1),YDOT,SIZE,23,90.0,-2)
467             X=(TIME(I)+TIME(I+1))/2-0.75
468             CALL SYMBOL(X,Y5,SIZE,'ANOX',0.0,4)
469       465   CONTINUE
470             CALL SYMBOL(1.5,YPLUS,1.0,'NITRATE',0.0,7)
471             CALL SYMBOL(1.0,Y2,SIZE,0,0.0,-1)
472             CALL SYMBOL(1.5,Y2,SIZE,'= NO3',0.0,5)
473             CALL SYMBOL(5.5,Y0,SIZE,'EFFLUENT',0.0,8)
474             CALL SYMBOL(5.0,Y1,SIZE,'THEO. EXP.',0.0,10)
475             CALL NUMBER(5.5,Y2,SIZE,ENO3,0.0,1)
476             CALL NUMBER(8.5,Y2,SIZE,AVNO3,0.0,1)
477             DO 565 I=1,NNO3,1
478             CALL SYMBOL(TNO3(I),PNO3(I),SIZE,0,0.0,-1)
479       565   CONTINUE
480             IF(NAMBA.EQ.1) GO TO 261
481             CALL SYMBOL(START,YFEED,SIZE,22,90.0,-1)
482             CALL SYMBOL(END,YFEED,SIZE,23,90.0,-2)
483             CALL SYMBOL(TEXT,YWRITE,SIZE,'FEEDING PERIOD',0.0,14)
484       261   CONTINUE
485             CALL LINE(T,NO3,NDH,1,0,0)
486             CALL PLOT (X2,0.0,-3)
487             CALL NAXIS(0.0,0.0,XAX,0,XSTUK,0.0,1,0.25,2)
488             CALL NAXIS(0.0,YAX,XAX,0,XSTUK,0.0,-1,0.25,2)
489             CALL NAXIS(0.0,0.0,YAX,0,YSTUK,90.0,-1,0.25,2)
490             CALL NAXIS(XAX,0.0,YAX,0,YSTUK,90.0,1,0.25,2)
491             DO 108 I=1,ITYD,1
492             TYD=T(1)+(I-1)*XAX*T(NDEL)/(ITYD-1)
493             IF(TYD.GT.24.0)TYD=TYD-24.0
494             XVAL=(I-1)*XAX/(ITYD-1.0)-0.5
495             IF(XVAL.LT.0.0) XVAL=0.0
496             CALL NUMBER(XVAL,-1.0,SIZE,TYD,0.0,-1)
497       108   CONTINUE
498             DO 103 I=1,IJ,1
499             YALK=(I-1)*YAX*ALK(NDEL)/(IJ-1)
500             YVAL=YALK/ALK(NDEL)
501             IF(YALK.LT.9.9) GO TO 701
502             IF(YALK.GT.9.9.AND.YALK.LT.99.9)GO TO 702
503             IF(YALK.GT.99.9)GO TO 703
504       701   CALL NUMBER(-0.5,YVAL,SIZE,YALK,0.0,-1)
505             GO TO 103
506       702   CALL NUMBER(-1.0,YVAL,SIZE,YALK,0.0,-1)
507             GO TO 103
508       703   CALL NUMBER(-1.5,YVAL,SIZE,YALK,0.0,-1)
509       103   CONTINUE
510             CALL SYMBOL(3.5,-2.0,SIZE,'TIME(HOURS)',0.0,11)
511             CALL SYMBOL(-2.0,6.0,SIZE,'ALK(PPM CACO3)',90.0,14)
```

```
512             DO 235 I=1,NFR,1
513             DO 235 J=1,KJ,1
514             Y=J-1.0
515             IF(TIME(I).GT.XAX) GO TO 235
516             CALL SYMBOL(TIME(I),Y,0.5,13,0.0,-1)
517         235 CONTINUE
518             DO 245 I=1,IFREQ,2
519             BEGIN=TIME(I)+SIZE
520             CALL SYMBOL(BEGIN,YDOT,SIZE,22,90.0,-1)
521             CALL SYMBOL(TIME(I+1),YDOT,SIZE,23,90.0,-2)
522             X=(TIME(I)+TIME(I+1))/2-0.75
523             CALL SYMBOL(X,Y5,SIZE,'ANOX',0.0,4)
524         245 CONTINUE
525             CALL SYMBOL(1.5,YPLUS,1.0,'ALKALINITY',0.0,10)
526             CALL SYMBOL(1.5,Y2,SIZE,'= ALK',0.0,5)
527             CALL SYMBOL(1.0,Y2,SIZE,0,0.0,-1)
528             CALL SYMBOL(5.5,Y0,SIZE,'EFFLUENT',0.0,8)
529             CALL SYMBOL(5.0,Y1,SIZE,'THEO. EXP.',0.0,10)
530             CALL NUMBER(5.5,Y2,SIZE,EALK,0.0,-1)
531             CALL NUMBER(8.5,Y2,SIZE,AVALK,0.0,-1)
532             DO 185 I=1,NALK,1
533             CALL SYMBOL(TALK(I),PALK(I),SIZE,0,0.0,-1)
534          185 CONTINUE
535             IF(NAMBA.EQ.1) GO TO 262
536             CALL SYMBOL(START,YFEED,SIZE,22,90.0,-1)
537             CALL SYMBOL(END,YFEED,SIZE,23,90.0,-2)
538             CALL SYMBOL(TEXT,YWRITE,SIZE,'FEEDING PERIOD',0.0,14)
539         262 CONTINUE
540             CALL LINE(T,ALK,NDH,1,0,0)
541             CALL PLOT(0.0,0.0,999)
542             STOP
543             END
```

```
      SUBROUTINE NAXIS(XST,YST,AXLEN,LOG,ANUMBR,ANG,LR,TICK,LINE)
C.... ROUTINE FOR DRAWING AXIS WITHOUT ANNOTATION
C.... (EITHER LOG OR LINEAR)
C....
C.... COMPLIMENTS OF YOUR FRIENDLY CHEM. ENG. DEPARTMENT.
C.... XST,YST = STARTING POSITION
C.... AXLEN   = LENGTH OF AXIS TO BE PLOTTED, CM
C....           (AXLEN MAY BE NEGATIVE)
C.... LOG     = STARTING POSN FOR LOG SCALE, USUALLY 1,
C....           OR = 0 FOR LINEAR SCALE
C.... ANUMBR  = NO. OF LOG CYCLES OR NO. OF TICK INTERVALS
C....           ALONG AXLEN.
C.... ANG     = 0. OR 90. FOR HORIZ. OR VERTICAL AXIS (ONLY)
C.... LR      = LOCATION OF TICK MARKS RELATIVE TO AXIS LINE:
C....           1 FOR CCW, -1 FOR CLOCKWISE.
C.... TICK    = SIZE OF TICK MARK, CM
C.... LINE    = 2 FOR SOLID, 3 FOR BLANK AXIS LINE:
C....           I.E., 3 WILL GIVE TICK MARKS ONLY.
      AXIS=0.
      DX1=0.
      DDX1=0.
      DX2=0.
      DY1=0.
      DDY1=0.
      DY2=0.
      LPOSN=LOG
      IF(AXLEN.NE.0.) GO TO 11
      WRITE(NPR,15)
   15 FORMAT(' WARNING - AXIS LENGTH IN NAXIS ROUTINE IS ZERO.')
      RETURN
   11 DELTA=AXLEN/ANUMBR
      IF(ANG.LT.10.) GO TO 1
      DX1=TICK
      IF(LR.GT.0) DX1=-DX1
      DDX1=DX1
      GO TO 2
    1 DY1=TICK
      IF(LR.LT.0) DY1=-DY1
      DDY1=DY1
C.... START AXIS
    2 CALL PLOT(XST,YST,3)
      X=XST
      Y=YST
    3 IF(LOG.EQ.0) GO TO 4
C.... SET UP LOG SCALE
      DELTA=(ALOG10(LPOSN+1)-ALOG10(LPOSN))*AXLEN/ANUMBR
      LPOSN=LPOSN+1
      IF(LPOSN.EQ.10) LPOSN=1
      DX1=DDX1
      DY1=DDY1
      IF(LPOSN.NE.2) GO TO 4
      IF(TICK.GT.(.1*AXLEN)) GO TO 4
      DX1=2*DX1
      DY1=2*DY1
    4 IF(ANG.LT.10.)DX2=DELTA
      IF(ANG.GT.10.) DY2=DELTA
      AXIS=AXIS+ABS(DELTA)
      CALL PLOT(X+DX1,Y+DY1,2)
      CALL PLOT(X,Y,3)
      IF(AXIS.GT.ABS(AXLEN)) GO TO 5
      X=X+DX2
      Y=Y+DY2
      CALL PLOT(X,Y,LINE)
```

```
64              GO TO 3
65        5     IF(ANG.LT.10.) X=XST+AXLEN
66              IF(ANG.GT.10.) Y=YST+AXLEN
67              CALL PLOT(X,Y,LINE)
68              RETURN
69              END
```